Jordan River DO TMDL Research Synthesis

Prepared for:

Utah Division of Water Quality
195 North 1950 West
P.O. Box 144870
Salt Lake City, UT 84114-4870
(801) 536-4300

Prepared by:

Cirrus Ecological Solutions, LC
965 South 100 West, Suite 200
Logan, UT 84321
(435) 787-1490

October 2020

CONTENTS

LIST OF TABLES

LIST OF FIGURES

ACRONYMS AND ABBREVIATIONS

AP	alkaline phosphatase
BCC	Big Cottonwood Creek
BG	β-1,4-glucosidase
BOD	biochemical oxygen demand
BODu	ultimate biochemical oxygen demand
BOM	benthic organic matter
C	carbon
CBOD	carbonaceous oxygen demand
Chl a	chlorophyll a
CPOM	coarse particulate organic matter
CR_{24}	24-hour community respiration
CVWRF	Central Valley Water Reclamation Facility
DO	dissolved oxygen
DOC	dissolved organic carbon
DON	dissolved organic nitrogen
DOM	dissolved organic matter
DWQ	Division of Water Quality
EEA	ecoenzyme activity
EEM	emission-excitation matrix
EPA	U.S. Environmental Protection Agency
FI	fluorescence index
FPOM	fine particulate organic matter
fDOM	fluorescent dissolved organic matter
G	grams
HMB	hydrologic mass balance
kg/yr	kilograms/year
Km	kilometer
JBWRF	Jordan Basin Water Reclamation Facility
LAP	leucine aminopeptidase
LJR	Lower Jordan River
m^2	square meter
mg/L	milligrams/liter
N	nitrogen
NAG	β-1,4-N-acetylglucosaminidase + LAP
O^2	oxygen
OM	organic matter
P	phosphorus

POX   phenol oxidase
QUAL2Kw   water quality model
SLM   simple linear mixing
SOD   sediment oxygen demand
SDSWRF   South Davis South Water Reclamation Facility
SVWRF   South Valley Water Reclamation Facility
TAC   Technical Advisory Committee
TP   total phosphorus
TDP   total dissolved phosphorus
TMDL   total maximum daily load
TOM   total organic matter
TSS   total suspended solids
UJR   Upper Jordan River
VSS   volatile suspended solids
WRF   water reclamation facility

1.0   INTRODUCTION

The Lower Jordan River (LJR; downstream of the 2100 South diversion) is currently impaired for low dissolved oxygen (DO), as well as several other parameters of concern including dissolved copper, *E. coli*, OE bioassessment, Total P, and total dissolved solids (Figure 1-1, DWQ 2016). The processes leading to low DO in the Jordan River have been analyzed repeatedly over the past 35 years (Stephens 1984, Jensen 1994, Borup and Smith 1999, Cirrus 2009, DWQ 2013). From a regulatory perspective, these processes are important to understand as they can be used to link pollutant sources to a water quality response.

Designated beneficial uses for the various segments of the Jordan River include domestic use (with prior treatment), secondary contact recreation (boating, wading, fishing, etc.), cold and warm water fisheries, other wildlife that depend on an aquatic environment (waterfowl, shorebirds, and the aquatic organisms in their food chains), and agricultural irrigation. These uses are protected by a variety of water quality standards, but every segment of the Jordan River has been found to be non-supporting of one or more beneficial uses (i.e., impaired) due to exceeding one or more water quality standards. Concentrations of DO are especially important to support the Class 3B beneficial use assigned to the LJR. Water quality standards associated with this beneficial use protect warm water fish species and other wildlife that depend on an aquatic environment (waterfowl, shorebirds, and the aquatic organisms in their food chains). DO in surface water is used by all forms of aquatic life and is a critical measure of water quality and ecological health.

Phase 1 of the Jordan River Total Maximum Daily Load (TMDL) was completed in June 2013 when the EPA approved the TMDL report (Division of Water Quality [DWQ] 2013). Although the report identified key parameters and processes that lead to impairment due to low dissolved oxygen (DO), the available data were insufficient to complete load allocations and reductions with the certainty desired by DWQ and other stakeholders in the watershed. As a result, the DWQ opted to pursue the phased approach to the TMDL. The U.S. Environmental Protection Agency (EPA) recommends the phased approach be "…limited to TMDLs that for scheduling reasons need to be established despite significant data uncertainty and where the state expects that the loading capacity and allocation scheme will be revised in the near future as additional information is collected" (EPA 2006).

The Phase 2 TMDL report is currently scheduled to be finished by fall of 2024. Since the conclusion of Phase 1, data collection and research have been completed that help address the data gaps and questions necessary to complete Phase 2. This work has been conducted by multiple academic, government, and private entities on water quality and ecological health of the Jordan River and tributaries to the river.

A previous research synthesis compiled and reviewed publications from 2010–2015 (Cirrus 2017). Research completed from 2015 to the present is reviewed in this current synthesis to determine data gaps from the 2017 synthesis that have been filled by more recent research and what gaps remain that may need to be filled to complete the Phase 2 TMDL. All research discussed in this synthesis contributes to an understanding of oxygen demand in the LJR by studying a process that consumes oxygen or by characterizing a pollutant source that influences such processes. What we looked for specifically was information that allows us to:

- Recommend a parameter of concern that links organic matter (OM), DO, and pollutant sources,
- Recommend options for quantifying differences in lability among OM sources, and
- Recommend methods for quantifying relative contributions of OM sources to sediment oxygen demand (SOD).

A full list of recent research that was considered for the synthesis is provided in Appendix A. This list comprises 42 peer-reviewed journal articles, unpublished research documents, and professional presentations that deal with some aspect of the Jordan River and the watershed area contributing to the river. Each was briefly reviewed to determine if it met the objectives of this synthesis discussed above and warranted inclusion in the synthesis. Some included useful information, but their results were described in

greater detail in other documents. Some provided supporting information and are referenced in the text as appropriate but not reviewed in detail. Others were not finished at the time but may eventually provide useful information. These are referenced in relevant sections of the synthesis to insure they are considered in the Phase 2 TMDL once completed. Finally, some documents were not included in the synthesis because they did not meet the objectives of the synthesis. All documents in Appendix A contribute to a greater understanding of the Jordan River and processes that influence the health of the river and the watershed.

The 16 documents carried into detailed review in the synthesis are shown in Table 1-1.

This synthesis is organized according to processes and sources that will be addressed in the Phase 2 TMDL. Figure 1-2, a diagram from the 2017 research synthesis, illustrates these variables. Oxygen demand is identified as the primary driver that influences chronic and acute DO impairment in the Jordan River. Oxygen demand can be divided between processes that occur in the water column and in the sediment. These processes can interact and contribute to impairment in different ways based on location and season. Recent research addressing oxygen demand is discussed in Section 2.

Processes that generate oxygen demand are influenced by pollutant sources that contribute OM loads. Characteristics of OM loads and processes that influence fate and transport of OM can determine how and when oxygen demand occurs in the LJR. Some of these characteristics include OM particle size (coarse, fine, and dissolved), labile or recalcitrant nature of OM, timing (seasonal, instantaneous, etc.) and location (upper and lower Jordan River). These characteristics are discussed in Section 3.

Section 4 of the synthesis describes recent computer modeling results. The model described in Section 4 is being developed for use in the Phase 2 TMDL to link pollutant sources with DO demand in the LJR. Load allocations in the Phase 2 TMDL will be supported in part by modeling results.

Section 5 provides a summary of key findings from the research that was reviewed in Sections 2 and 3. These results are combined with key findings obtained from the 2017 synthesis and organized to identify how information from each synthesis will support Phase 2 TMDL requirements. Section 5 concludes with a list of data gaps in the current understanding of DO demand and OM processes in the LJR and provides recommendations for filling the gaps. DWQ will determine which of these gaps should be filled prior to or during the Phase 2 TMDL.
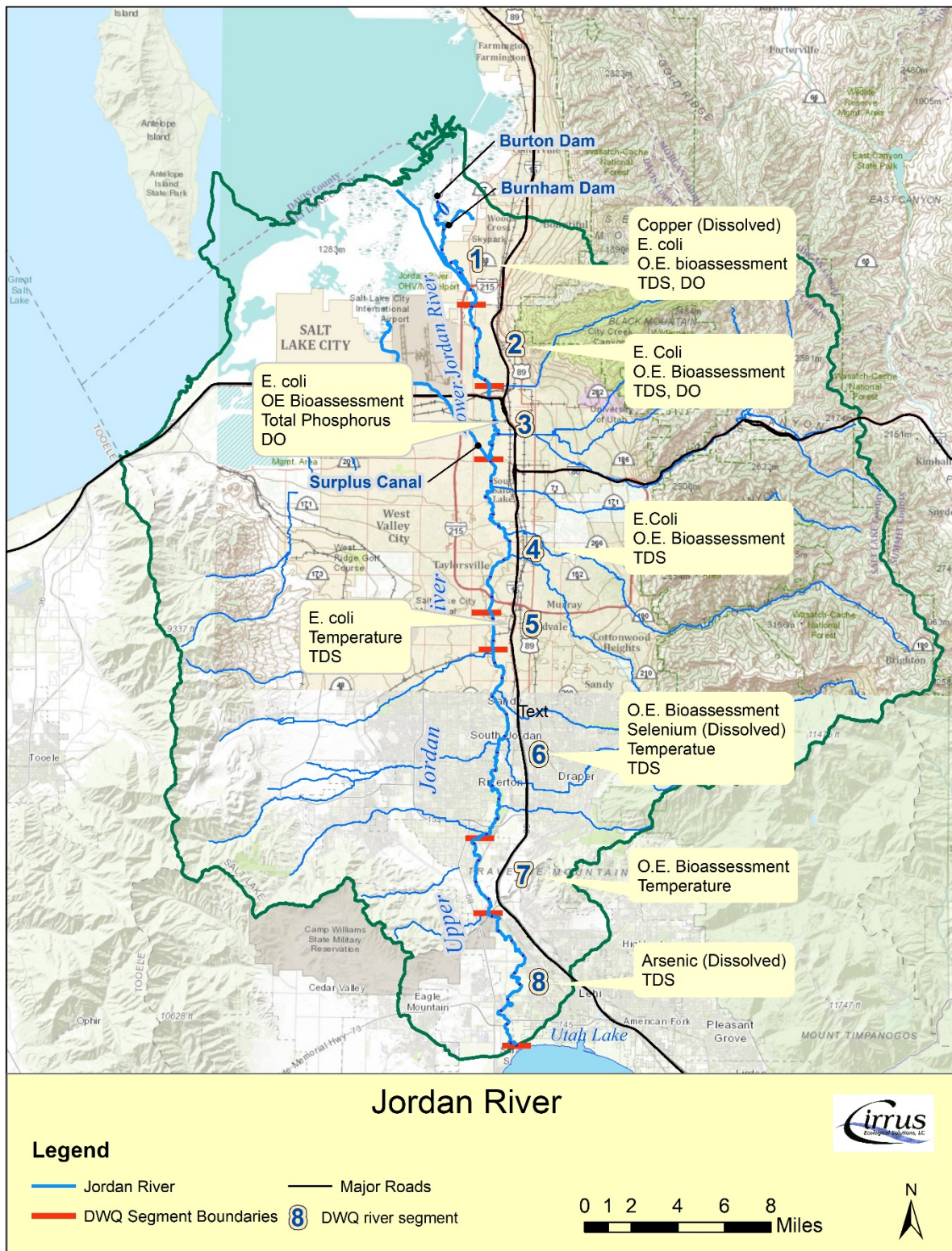
Figure 1-1. Jordan River segments and existing water quality impairments based on most recent integrated report (DWQ 2016).

Table 1-1. Publications reviewed in detail in the Jordan River DO TMDL Research Synthesis.		
Author	**Year**	**Title**
Cirrus Ecological Solutions, LC	2016	Lower Jordan River: Phase 2 Report. Dissolved Oxygen Response to Flow Change (2011–2016).
Cirrus Ecological Solutions, LC	2017	Lower Jordan River Diurnal Dissolved Oxygen Assessment (July 29–August 4, 2017).
Dupont et al.	2018	Demonstrating the Pollutant Loading from Stormwater Discharge to an Urban River in the Intermountain West Using High-Frequency Data.
Follstad Shah et al.	2017	Nitrogen Sources and Transformations within the Jordan River, Utah, and Microbial Community Response to Energy and Nutrient Availability in the Jordan River, Utah.
Follstad Shah et al.	2018	Linking Water Sources and Water Quality within the Jordan River, Utah.
Follstad Shah et al.	2019	Microbial Community Response to Shifting Water Quality and Quantity in an Arid Urban Ecosystem.
Follstad Shah et al.	2019	Spatiotemporal Variability in Water Sources Controls Chemical and Physical Properties of a Semi-arid Urban River System.
Goel and Abedin	2016	Nitrogen Dynamics in the Jordan River and Great Salt Lake Wetlands.
Kelso, J.	2018	Organic Matter Sources, Composition, and Quality in Rivers and Experimental Streams.
Khatri et al.	2019	Impact of Climate and Land Use Change on Streamflow and Sediment Yield in a Snow-dominated Semiarid Mountainous Watershed.
Miller, T.	2019	Chapter 2: Light Attenuation and Nutrients in the Jordan River and Their Relationship to Periphyton and Phytoplankton Communities.
Miller, T.	2019	Chapter 3: Coarse Particulate Organic Matter Measurements in the Jordan River and its Major Tributaries, 2010–2015.
Miller, T.	2019	Chapter 4: Diel Patterns of Dissolved Oxygen, Flourescing Dissolved Organic Matter and Turbidity, and Their Relationship to Seasonal Runoff and Storm Events in the Jordan River.
Richards, D.C.	2018	Chapter 12: A Snail, a Clam, and the River Jordan: A Revealing Novel.
Salt Lake County Watershed Planning and Restoration Program	2018	Section 319. Nonpoint Source Pollution Control Program Watershed Project Final Report.
Su, J.	2019	Jordan River Modeling Through the Water Quality Assessment Simulation Program (WASP): Model Report.
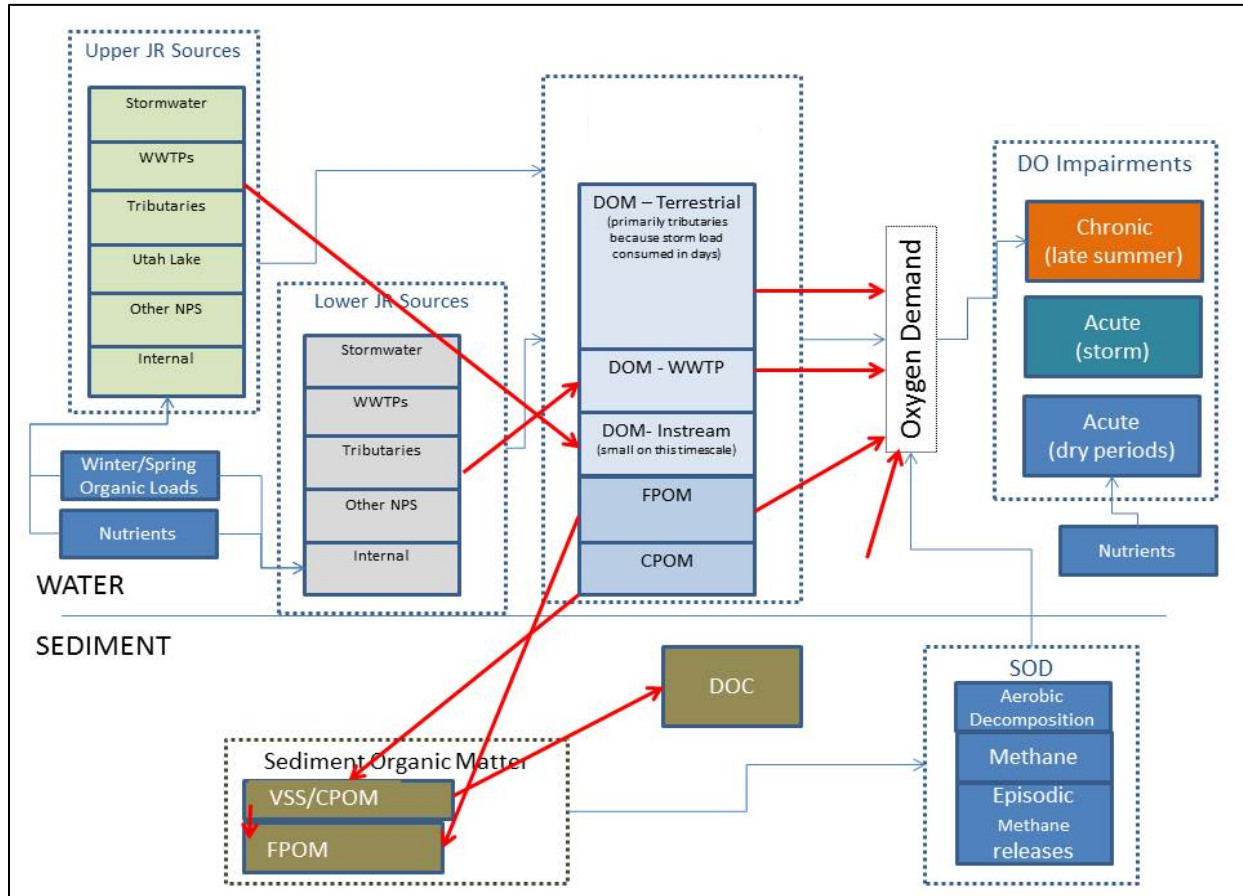
Figure 1-2. Jordan River Total Organic Matter (TOM) and DO linkages based on current understanding and regulatory needs of the Phase 2 TMDL. Source: Cirrus (2017a) Figure 15.

This page is intentionally left blank.

2.0   DISSOLVED OXYGEN DEMAND

As noted above, DO demand is associated with processes that occur in the water column and processes that occur in the sediment. The Phase 2 TMDL will need to link DO demand processes in each category to pollutant sources. Load allocations may then be developed to account for the influence that each source has on DO demand.

At the time the Phase 1 TMDL was completed, most evidence indicated that DO demand in the Jordan River was influenced by four major processes including algal productivity/respiration (i.e., stream metabolism), aerobic decomposition in the water column, physical processes (i.e., reaeration), and aerobic decomposition/inorganic oxidation in sediment (Cirrus 2009). Research completed in the past 10 years has provided additional detail on some of these processes and how they influence DO demand.

This section first summarizes relevant findings of the Phase 1 TMDL and the 2017 synthesis (Sections 2.1 and 2.2) then discusses the results of recent research (2015 to present) that build on our previous understanding of DO demand (Sections 2.1.1–2.1.4 and 2.2.1–2.2.2).

2.1 DO DEMAND IN THE WATER COLUMN

Based on available data, the Phase 1 TMDL proposed that algae growth and productivity were limited in the LJR by turbid conditions created by suspended solids and less so by nutrient concentrations. This interpretation was based on monitoring data that indicated a decrease in the diel range of DO between 2100 South and Cudahy Lane. No information was identified in the previous synthesis to conclude if this assumption was correct. The range of diel DO concentration measured below water reclamation facilities (WRFs) did not seem to be significantly influenced, indicating a limited response by algae to this source of nutrients, although groundwater upwelling (containing low DO) has been hypothesized as a potential influence on the diel range (DWQ 2015a). Limited algal monitoring data were available at the time to compare diel DO or other parameters (e.g., algal samples, Chl a, etc.) at different locations and between seasons or years. It may be expected that diurnal variations in groundwater discharge to the river could result in some impacts on DO due to changes in temperature-based viscosity and evapotranspiration. However, these impacts would be minor with expected higher DO concentrations resulting from groundwater inflow to the Jordan River (Thiros 2003).

The previous synthesis compared estimates of stream metabolism (DWQ 2015b, based on algal productivity and respiration) to direct measurements (Hogsett 2015). Results of these studies were in general agreement and concluded that upper Jordan River segments are net autotrophic and LJR segments are net heterotrophic when considering total DO demand (i.e., water column and benthic metabolism together; Epstein et al. 2016).

The Phase 1 TMDL primarily used measurements of biochemical oxygen demand (BOD) to characterize oxygen demanding processes in the water column. Additional samples that indicate oxygen demand (e.g., BOD, carbonaceous oxygen demand [CBOD], etc.) were not available for the previous synthesis to compare with data used in the Phase 1 TMDL. The synthesis recommended use of dissolved organic carbon (DOC) as a monitoring parameter for water column oxygen demand, based on results that indicated most demand is generated by biodegradable carbon and not nitrogen (N).

The synthesis also reported that direct measurements of water column oxygen demand were greatest in upper Jordan River segments (Hogsett 2015) and that demand decreased dramatically during the winter season (Hogsett 2015). Seasonal increases in water temperatures and warm effluent discharge during the winter were considered to increase water column metabolism.

No information was reported on other factors that influence microbial response (e.g., nutrients or carbon availability). Continuous measurements of fluorescent dissolved organic matter (fDOM) represent the dissolved component of OM in the water column. Available diel monitoring indicated that fDOM and DO

had an inverse relationship and that storm events showed measured increases in fDOM, particularly during the spring season (Cirrus 2017).

Measured reaeration rates (Hogsett 2015) were compared in the previous synthesis to measured and modeled rates used in the Phase 1 TMDL. Rates from both sources showed a general decrease with distance downstream in the Jordan River, corresponding to reductions in stream-channel gradient and increased stream depth. Based on measured results, low reaeration rates are a concern at times in the LJR. The synthesis reported that reaeration created by the 2100 South diversion contributed approximately 0.3 milligrams/liter (mg/L) DO as water passed through the radial gates (Salt Lake City 2015). Computer modeling indicated that flow increases of 25 percent above baseflow at 2100 South would increase DO levels in the LJR above the 5.5 mg/L chronic standard (SWCA 2014). Use of engineered structures has also been reviewed, indicating that mechanical reaeration could temporarily improve DO during the fall season when chronic DO impairment is greatest (Aqua Engineering 2013). On average, 1.3 mg/L of DO would need to be added mechanically (approximately 42 lb/hr) to meet the instantaneous minimum DO standard of 4.5 mg/L. However, oxygen demand generated during storm events would require substantially more oxygen (and associated cost) to offset the demand, making mechanical reaeration financially unfeasible at this time.

2.1.1 Algal Growth and Productivity

Light availability is a critical factor for algae growth in the water column and on channel substrate. Other important factors include a source of algae and bioavailable nutrients, which are supplied to the Jordan River by Utah Lake and WRFs, respectively. Benthic algae must also have stable substrate that remains free of sedimentation to colonize and sustain a viable population.

Light availability measurements (July–October 2009) from 22 sites on the Jordan River and Surplus Canal showed rapid attenuation with depth but identified sufficient light (>5 percent surface irradiance) at the channel floor at most sites to support growth of benthic algae (Miller 2019a). Sites that did not have sufficient light to support algal growth at full depth included several downstream sites at Center Street (Cudahy Lane), 500 m below South Davis South Water Reclamation Facility (SDSWRF), 500 m below Burnham Dam, and the Surplus Canal.

Measurements of water column algae (June–October 2009) collected from the same locations as light attenuation indicated that plankton flora in the Jordan River is not resident but largely originates from Utah Lake and from periphyton on the bottom of the river (see Rushforth and Rushforth 2009a, p. 6). Large reductions in the level of Chl a and biovolume of algal species were observed due to the Turner Dam diversion (Figure 2-1).

Any change in biovolume below Turner Dam for a given month or between months could not be determined due to the scale used to present data in Miller (2019a). Individual measurements should be reviewed for each species at each monitoring location to determine the potential for algal settling in the LJR. Additional analysis of phytoplankton and benthic algae biomass measurements could provide additional evidence regarding which source is the dominant driver of productivity in the Jordan River.

Although nutrient concentrations in summer 2010 for middle and lower Jordan River segments ranged 0.10–0.65 mg/L for Total P and 0.30 – 1.1 mg/L for Total N, a corresponding increase in algal biovolume and Chl a was not observed downstream of monitoring sites. Regression analysis of monitoring data indicated no significant relationship between nutrient concentrations and measurements of algae or periphyton (Miller 2019a). However, a study of nutrient dynamics in the Jordan River measured simultaneous negative fluxes of nitrate and phosphate at 1300 South and at the Legacy Nature Preserve (Goel and Abedin 2016). Evidence of instream nutrient processing is also suggested by differences in the measured and cumulative total dissolved phosphorous (TDP) load between South Valley Water Reclamation Facility (SVWRF) and Central Valley Water Reclamation Facility (CVWRF; Figure 2-2,

Follstad Shah et al. 2018). Although some uncertainty still exists, nutrient flux response and nutrient processing can be explained in part by algal uptake or microbial activity.
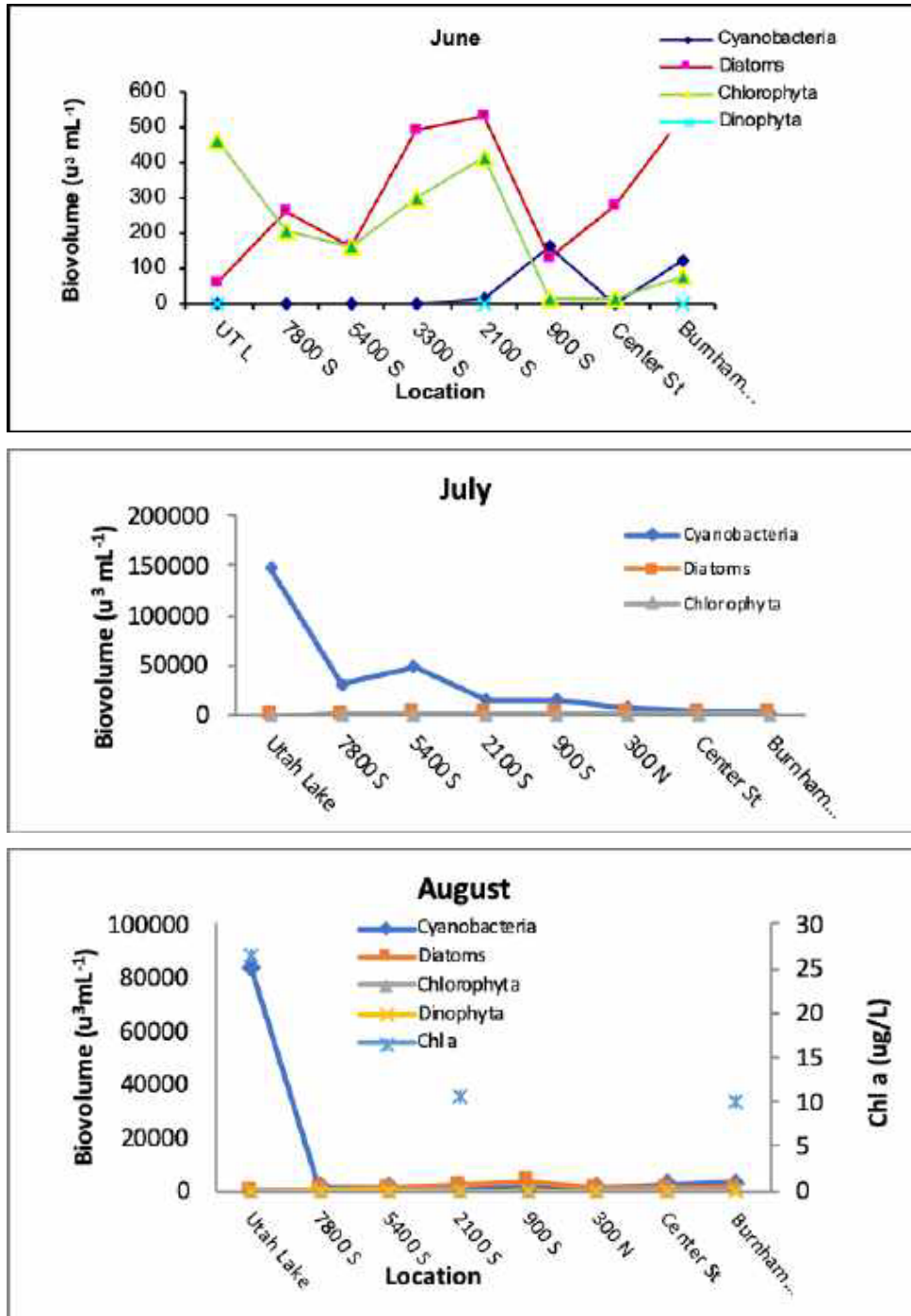
Figure 2-1. Measurements of algal species at select sites in the Jordan River measured during June, July, and August 2009. Note large reductions in biovolume between Utah Lake and 7800 South due to the Turner Dam diversion. Source: Miller (2019a) Figure 21b.
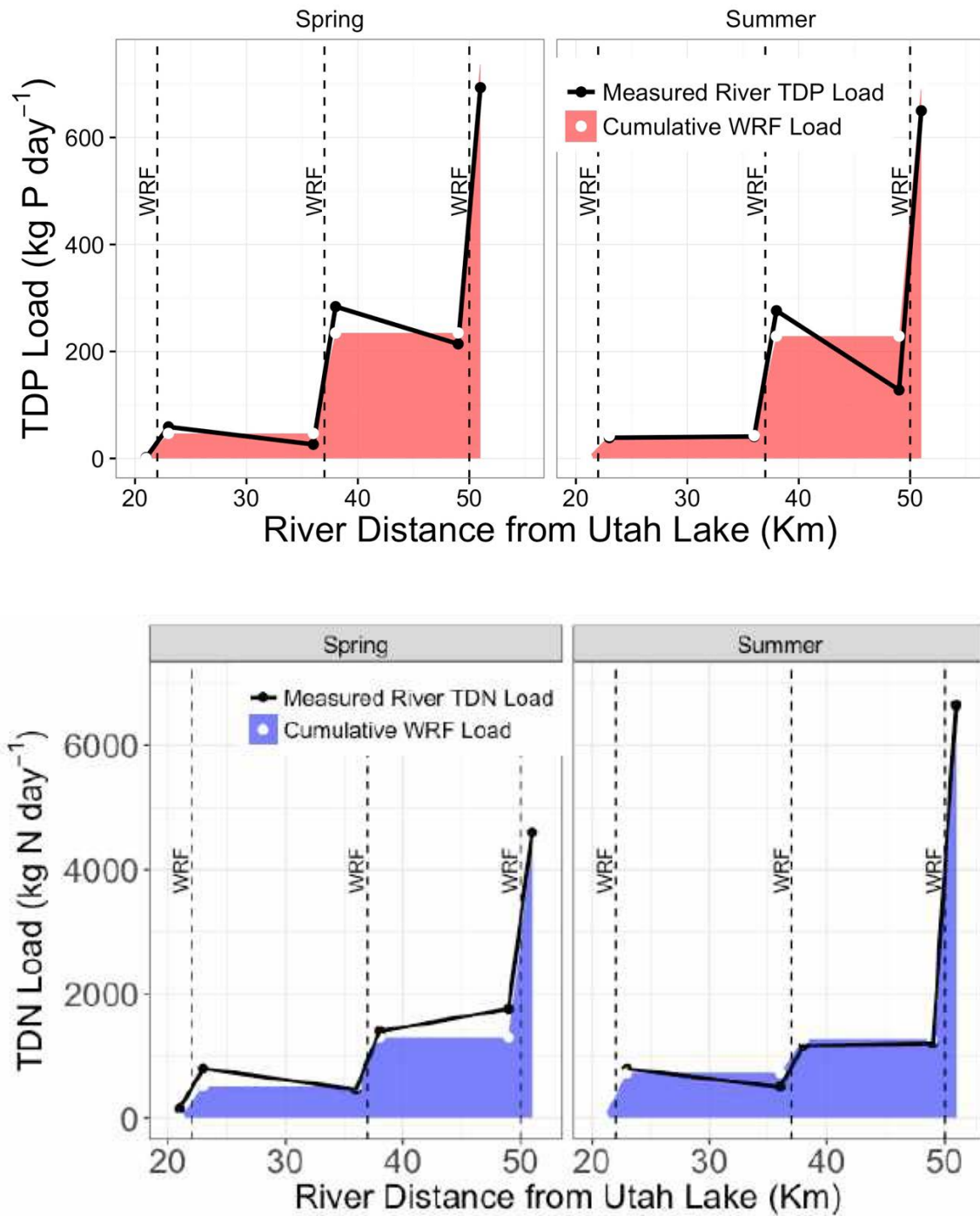
Figure 2-2. Phosphorus and nitrogen load analysis measured in spring (late May) and summer (mid-August) 2016 from the Jordan River. Evidence of instream processing occurs when measured river nutrient load (solid line) is less than the cumulative nutrient load (shaded areas) from effluent inputs. Processing mechanisms could be in part due to microbe consumption or algal uptake. Source: upper plot - Follstad Shah et al. (2018), lower plot – Follstad Shah et al. (2017).

Benthic algae measurements were collected on seven sample dates June–October 2010 at light-attenuation monitoring sites using slate tiles either placed horizontally on the channel bed or vertically fixed in the water column or from available cobble substrate (Miller 2019a). Results show that peak Chl a was consistently measured in segments below 7800 South when considering all three substrate types and sample dates (Miller 2019a). Downstream of 7800 South, the Jordan River receives nutrient input from four WRFs before reaching Burnham Dam, which indicates potential for some influence on algal growth. Chl a measured at sites below effluent discharge points both decreased and increased depending on location and substrate type. Chl a measurements were consistently higher on suspended tiles compared to bottom tiles that were covered with sediment deposits at some locations. Repetitive scouring and deposition in response to highly variable flows during the 2010 survey appeared to significantly impact the opportunity for benthic algal growth in the LJR. Loss of benthic algae could also occur through grazing by benthic invertebrates and fish benthic algaevores.

A water quality response to algal productivity can be monitored using diel DO measurements. Typical diel DO cycles peak near the end of the photosynthetic period when incoming solar energy is greatest. Diel DO measurements in late June 2013 at 2100 South showed a wide range from 5.5–9.5 mg/L along with an increasing range in pH, which indicates increased biological activity (Miller 2019b). Over time, peak DO remained consistent while minimum DO continued to sag (Figure 2-3). This pattern occurred at downstream LJR sites as well. The pattern suggests respiration has a greater influence on the daily DO balance than photosynthesis or reaeration, particularly at LJR sites (Miller 2019b).

Diel DO cycles measured during summer 2017 were compared at nine locations in the LJR (Cirrus 2017b). The results showed an odd shift in the diurnal pattern of DO from site to site (Figure 2-4). Review of diel DO during the winter season showed this shift is absent. A typical DO pattern (influenced by algal photosynthesis and respiration) has a maximum DO sometime past solar noon, which occurs after 1:30 p.m. during July and August for the LJR. Measurements at the upper three LJR sites showed this typical diurnal DO pattern. However, the three middle LJR sites had peak DO during times when photosynthesis is absent and DO should be at a minimum level. This anomaly was still evident at Cudahy Lane and Burnham Dam, which showed peak DO in the mid-morning when photosynthesis is typically increasing, following a nightime period dominated by respiration. Plotting the travel time of 150 cfs with the delta time in DO peak showed a similar pattern in timing. This suggests that during stable periods, the diurnal pattern of DO in the LJR is dependent upon DO at 2100 South. It also suggests that peak DO in the LJR is driven not by photosynthesis but by physical processes such as transport of DO from upstream and/or temperature. In other words, primary productivity in the UJR sets the minimum daily DO entering the LJR, which is then lowered during the summer months through decomposition of OM.

2.1.2 Aerobic Decomposition in the Water Column

Oxygen is consumed when OM is decomposed in the water column by microbial action. Growth of microbial communities in the Jordan River depends largely on dissolved organic matter (DOM) contributed by various sources. The influence of microbial activity on DO is represented with measurements of BOD or CBOD. These parameters were measured monthly 2009–2012 to characterize seasonal and annual variability (Miller 2019b).

Annual BOD averages at most sites during 2009–2011 ranged from 2.0 to 5.0 mg/L with some exceptions outside this range occurring during 2010 (a wet year) and at sites downstream of Utah Lake or below WRFs. Measurements reported in Miller (2019b) are assumed to represent values from a 5-day test although this was not explicitly stated. An assessment of average concentrations based on 2012 data did not identify consistent differences during November–April and May–October (Figure 2-5). A more detailed look at all measurements is recommended to determine if temporal patterns exist at finer scales during a given year or consistently across several years.
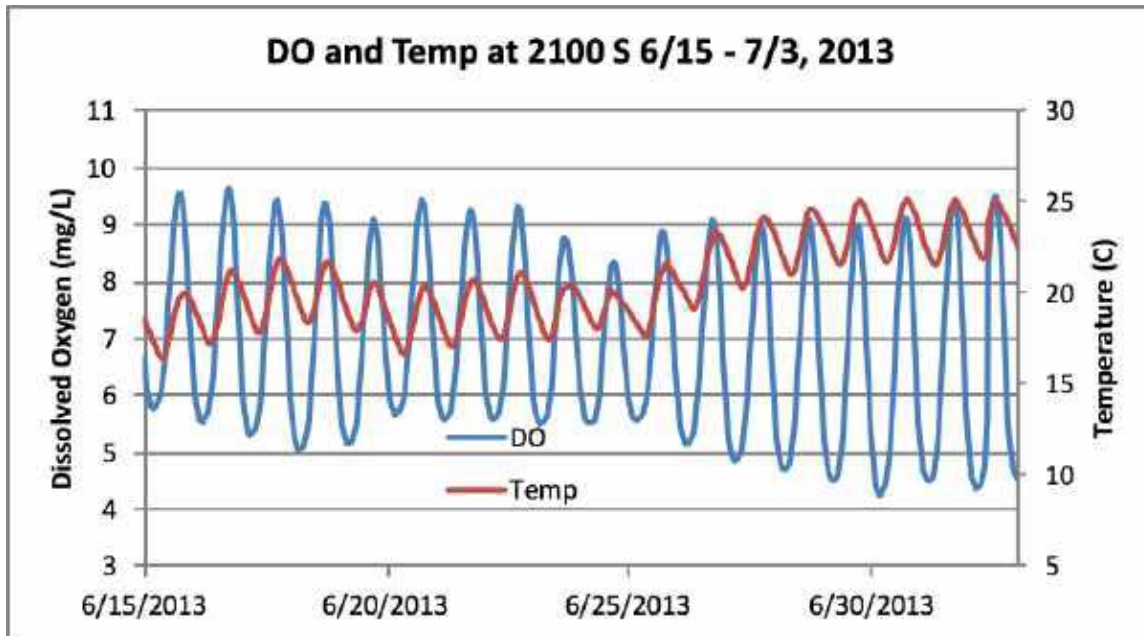
Figure 2-3. Diel measurements of DO and temperature at collected from the Jordan River at 2100 South, June 15–July 3, 2013. Source: Miller (2019b) Figure 90b.
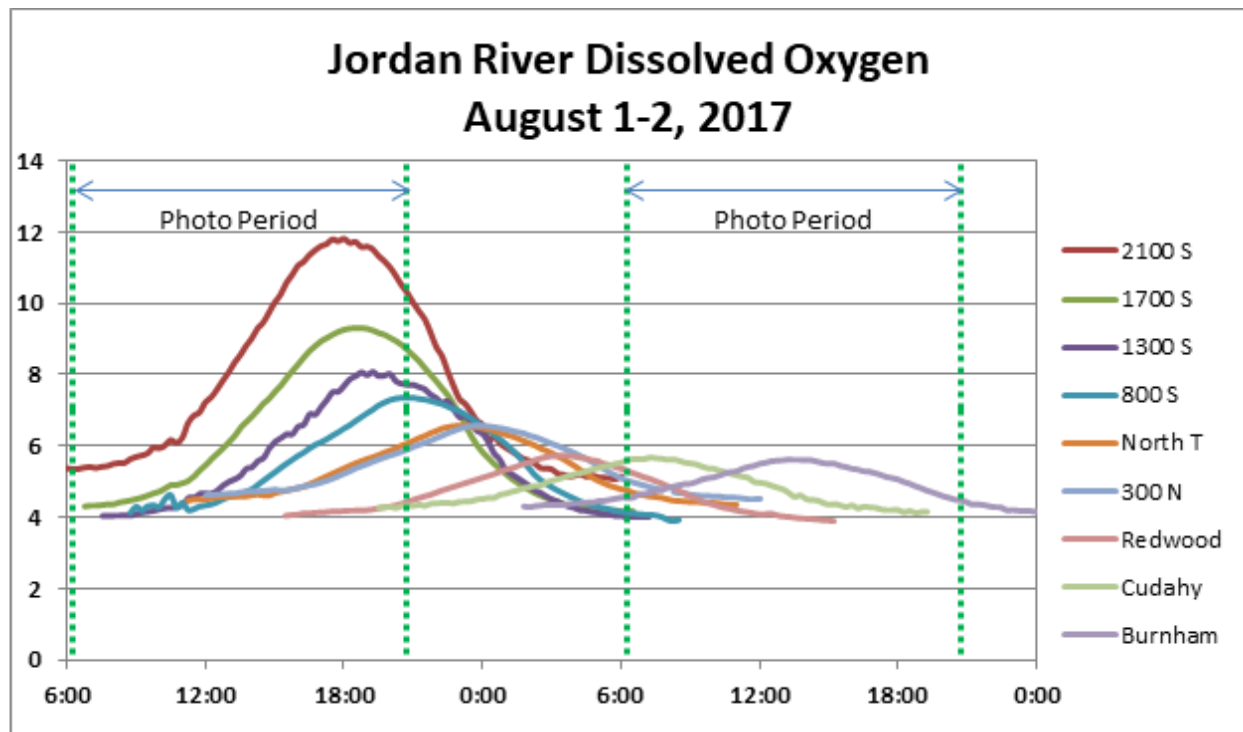
Figure 2-4. Dissolved oxygen diel measurements for a 24-hour period starting at the DO minimum value at 2100 South illustrating the temporal shift in DO that occurs in the lower Jordan River. Note that peak photosynthesis occurs outside the photoperiod beginning at 800 South and extending downstream through Redwood Road. Source: Cirrus (2017b) Figure 3.
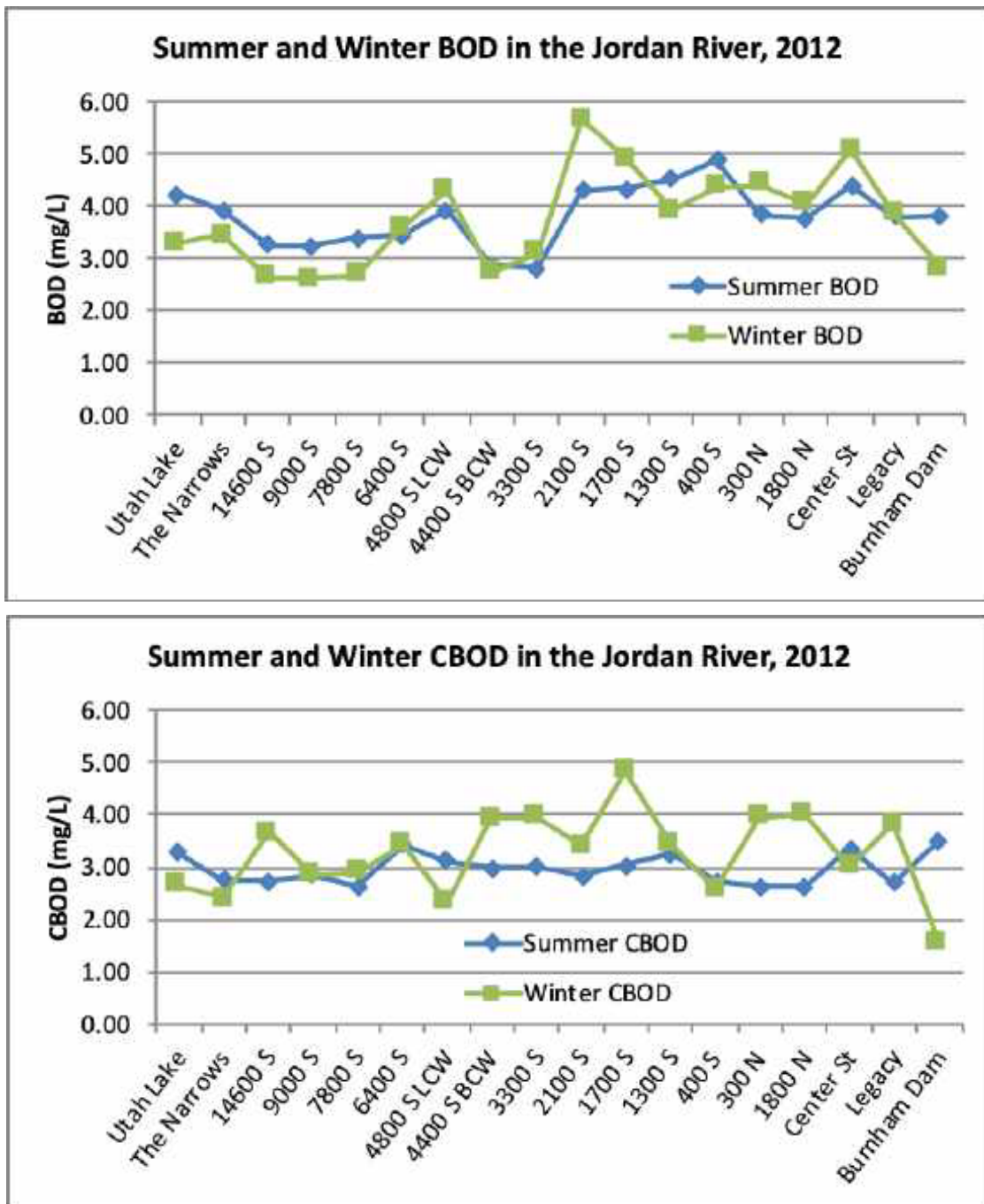
Figure 2-5. Seasonal assessment of BOD and collected from select locations on the Jordan River during 2012. Fifteen samples were collected from each site and included at least one sample each month. Results show seasonal average concentrations for summer (May-October) and winter (November-April). Source: Miller (2019b) Figure 66.

The process of OM decomposition is facilitated by bacteria that secrete enzymes into the environment to degrade complex organic compounds. The balance of enzymes secreted by bacteria to acquire energy or nutrient resources depends on the availability of C, N, and P in the environment relative to the stoichiometric balance of C, N, and P within microbial biomass.. Decomposition of OM is considered a

significant rate-controlling step in the global carbon cycle. As a result, substantial research has occurred over time on enzyme reactions, from small to large environmental scales. Enzyme research has traditionally addressed activity in either soil (Burns 1978) or aquatic resources (Overbeck 1991). The focus of soil and aquatic research eventually converged as a result of advances in methods and technology. In the more recent past, studies of ecoenzyme activity (EEA) and simulation models have connected EEA to ecological process and theory (Sinsabaugh and Follstad Shah, 2012).

The way microbial communities perceive energy and nutrient resources in the environment can be inferred from EEA. Ecoenzymes are enzymes produced by microbes but are not bound by cell membranes. They are expressed by bacteria to obtain nutrients and energy from OM, including N, P, and C. Codes that represent ecoenzymes produced by microbes include LAP (leucine aminopeptidase), NAG (β-1,4-N-acetylglucosaminidase + LAP), AP (alkaline phosphatase), POX (phenol oxidase), and BG (β-1,4-glucosidase, Sinsabaugh and Follstad Shah 2012). These enzymes are used to acquire N from proteins and other polypeptides (LAP), cleave glucosamine from chitin and peptidoglycan to acquire Ca and N (NAG), hydrolyze P from phospholipids, phosphosaccharides, and nucleic acids (AP), catalyze the oxidative degradation of polyphenols (e.g., lignin, tannins) and hydrocarbons (e.g., humic substances) to acquire more recalcitrant C (POX), and hydrolyze glucose from polysaccharides (e.g., cellulose found in plant tissue or algal cells walls) to acquire more labile C (BG).

When resource supplies are stable, microbial communities are eventually co-limited by C, N, and P, and ecoenzymes are expressed in a 1:1 ratio with each other. Expression of ecoenzymes changes with resource availability and OM quality. For example, N and P fertilization in urban settings may increase BG activity and decrease LAP, AP, and POX activity.

Ratios of BG:LAP and BG:AP measured in the Jordan River water column are within the range of values for rivers from published literature, although they indicate an imbalance (<1) for N and sometimes (<1) for P, which means either that more units of nutrients are needed per unit of C to sustain microbial metabolism and growth, or that microbial communities are investing in greater acquisition of abundant N and P in organic forms (Figure 2-6).

Seasonal measurements of EEA rates (umol/L-hr) for AP, BG, and LAP were collected during baseflow conditions above and below effluent inputs in 2016 (Follstad Shah et.al. 2019a). Results indicated that BG and LAP were generally stable with high activity rates over the length of the river. AP activity was highly variable and switched between generally high rates in either the upper or lower Jordan River segments, based on season. Given the distribution and location of WRFs, these responses do not follow global ecoenzyme relationships based on resource availability (Sinsabaugh and Follstad Shah 2012). Regression analysis of Jordan River samples indicated that EEA rates were uncorrelated with each other. Uncoupled ecoenzyme activities are likely the result of large inputs of N, P, and C. Furthermore, regression analysis results did not change between season, indicating that uncoupling is not related to seasonal variation in resource availability (Follstad Shah et.al. 2019b).

Analysis of limitations on microbial growth show conflicting results (Figure 2-7). Based on microbial biomass C:N:P (60:7:1) and dissolved forms of organic C, N, and P in the water column and effluent, microbial growth in the Jordan River water column is predicted to be co-limited by C and P, relative to N. Based on measured ecoenzyme ratios related to C:N:P acquisition (1:1:1), microbes are N-limited. However, this interpretation is based on a paradigm largely built upon studies in nutrient-limited terrestrial systems. Organic N is in ample supply in the Jordan River (Figure 2-7). The greater investment in LAP relative to BG therefore suggests that microbes in the water column are utilizing abundant, protein-rich DOM, including algae, other bacteria, and dissolved biosolids. Regression analysis of ecoenzyme activities show consistent positive relationships between C and N (LAP + NAG) acquisition with slopes <1 (Follstad Shah et al. 2017), but no correlation exists when LAP activity alone is regressed against BG activity (Follstad Shah et al. 2019b). This difference can be attributed to NAG production targeting chitin and peptidoglycan resources, which are rich in both C and N. Chitin is a primary component of fungal cell

walls, arthropod exoskeletons, and mollusk shells, while bacterial cell walls are comprised of peptidoglycan.

Ecoenzyme analysis also suggests that microbial communities in the water column can perceive differences in resource supply based on current metabolic needs and respond most to available P resources in comparison to C or N (Follstad Shah et al. 2017). This was evident during summer when results showed increased P acquisition, likely in response to higher growth rates supported by cell structures rich in P, such as ribosomes. As previously discussed, measurements of AP activity showed spatial and temporal variability. However, measurements consistently showed higher rates downstream of older WRFs during October and periodically in August and May (Follstad Shah et al. 2019b).

Although current results apparently conflict with traditional controls on microbial activity, consumption of water column OM remains a primary concern regarding DO demand. Previous research indicates DOM comprises the majority of water column OM in the LJR system (Epstein et al. 2016). This large OM contribution results in net heterotrophy in those segments and fuels metabolic consumption of oxygen. Further, Follstad Shah et al. (2017), and Kelso and Baker (2020) showed that this DOM is largely microbially produced, with very high fluorescence index (FI), implying the DOM is highly available for microbial consumption. Measurements of EEA provide additional evidence of the labile nature of Jordan River DOM (discussed below in section 3.3) Therefore, future consideration of chronic low DO during dry (non-storm) conditions should continue to identify factors influencing the production of protein-rich DOM substrates that increase microbial activity and generate oxygen demand.

2.1.3 Macroinvertebrate Influence on Water Quality

Macroinvertebrates are used by Utah DWQ to indicate water quality and stream system health. Measurements of invasive macroinvertebrates in the Jordan River including the Asian clam (*Corbicula fluminea*) and New Zealand mudsnail (*Potamopyrgus antipodarum*) have been noted in the literature to occur at levels with potential to influence water quality (Richards 2018). This section reviews recent information collected on these two species in the Jordan River.

Corbicula prefer to live in well-oxygenated sediments and appear in much lower densities where sediments have high organic content and levels of low oxygen. They can grow from a juvenile of 1 mm to a 5-cm adult within a period of 1–4 years and create shells that are approximately 12 percent C by weight. The clam consumes POM from the water column and sediment then excretes nutrients in dissolved form or as feces or pseudofeces. *Corbicula* typically consume POM in the range of 1–20 um but can consume spherical algae with a diameter from 50 um up to 170 um.

Rough estimates of densities of the *Corbicula* in run habitat between 5400 South above the Mill Creek confluence downstream to 900 South range from 175–2,635 live clams/m^2 (square meter) and four live clams/m^2 in pool habitat. Densities of the *Potamopyrgus* are estimated at 500,000/m^2 in this same reach, based on literature values for a highly productive stream (Richards 2018).

Based on literature values of filtering rates and estimates of density, *Corbicula* could potentially filter large volumes of the Jordan River depending on the estimated density of this species (Richards 2018). In spite of the increased oxygen demand created by *Corbicula* the net benefit of this macroinvertebrate could potentially be positive if sufficient OM was consumed to reduce turbidity and establish benthic algae along with photosynthetic oxygen production (Figure 2-8). The contribution of macroinvertebrates to oxygen demand in the water column and sediment has been measured in combination with other processes during earlier research of SOD (Hogsett 2015). Additional information on sediment processes affected by macroinvertebrates is discussed below in Section 2.2.2.
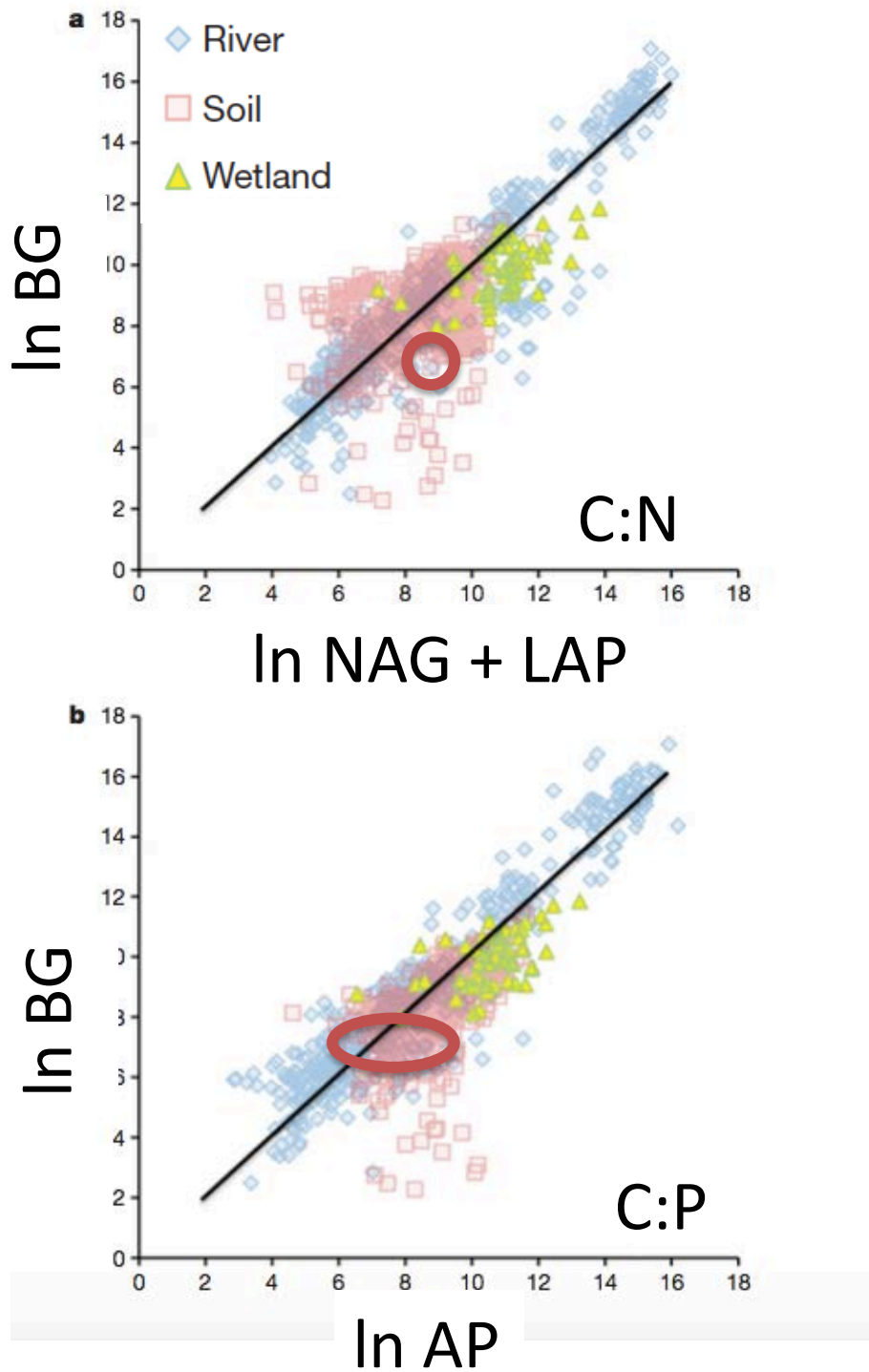
Figure 2-6. Ratios of ecoenzymes used by microbes to obtain C (BG), N (NA + LAP), and P (AP) resources from organic material. Note the upper plot shows C:N ratios while the lower plot shows C:P ratios. Shaded symbols represent ratios measured from river, soil, and wetland samples. Circled areas represent the range of sample measurements collected from the Jordan River. Source: First published in Sinsabaugh et al. 2009; shown here from Follstad Shah et al. (2019b).

WATER CHEMISTRY STOICHIOMETRY
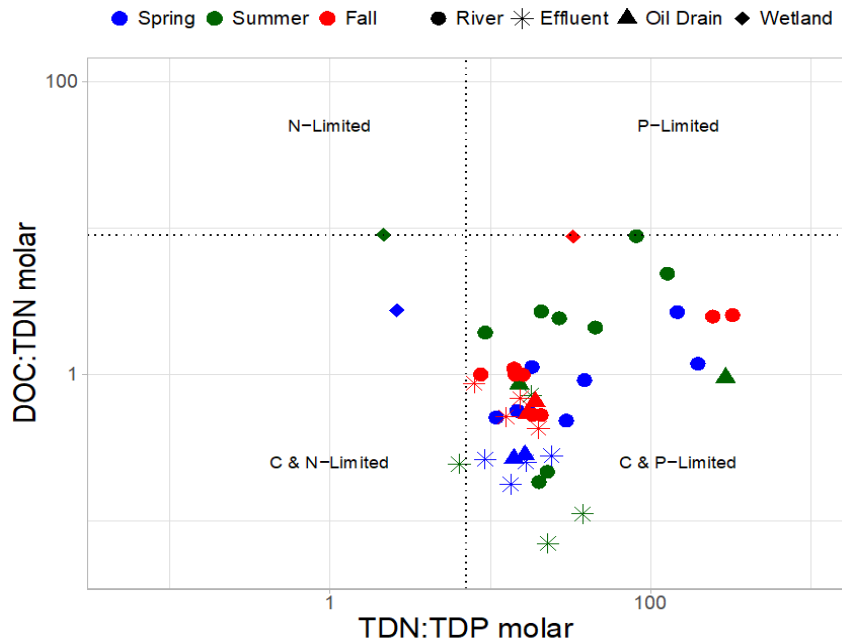
Thresholds based on microbial C:N:P (60:7:1)

ECOENZYME STOICHIOMETRY
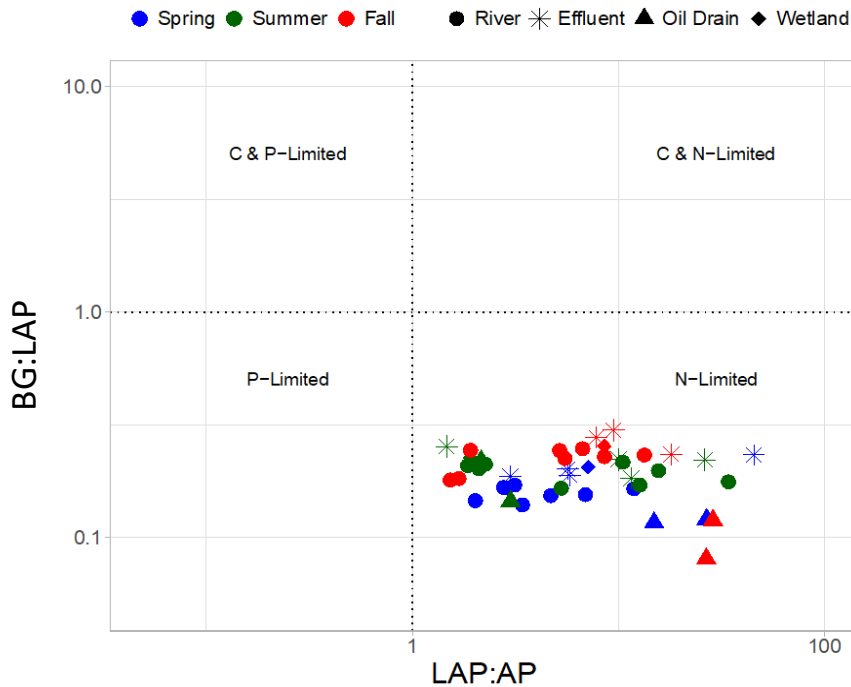
Thresholds based on global ecoenzyme ratios related to C, N, & P acquisition (1:1:1)

Figure 2-7. Resource limits on microbial growth determined from water quality stoichiometry (C and P-limited) and ecoenzyme stoichiometry (N-limited). Source: Follstad Shah et al. (2019b).
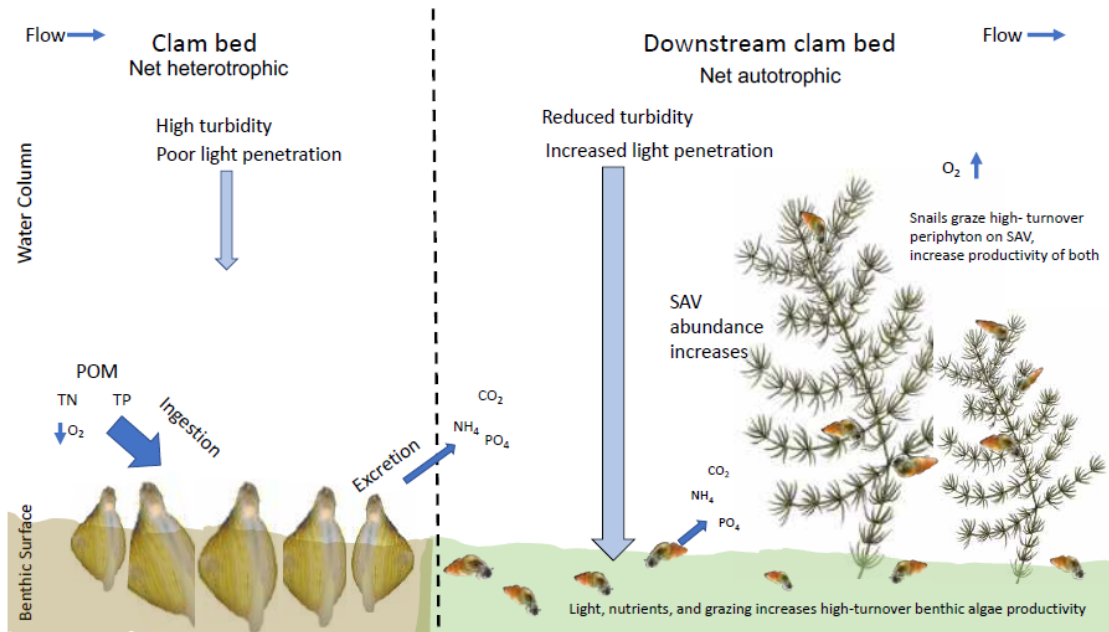
Figure 2-8. Potential synergistic effects of *Corbicula* and *Potamopyrgus* on nutrient cycling in the Jordan River. Source: Richards (2018) Figure 67.

2.1.4 Hydrology and Oxygen Demand

Water column oxygen demand in rivers can be influenced by the quantity and quality of flow. Variations in flow and water quality are common in urban stream systems. Isotope tracing was used to develop a mass balance hydrology model for the Jordan River to quantify when and where major shifts in source water occur (Follstad Shah et al. 2018, Follstad Shah et al. 2019a). A total of 18 study sites were established along the Jordan River from Utah Lake downstream to near the Redwood Road Crossing at 1800 North. Flow and water quality measurements, including stable isotopes ($\delta^{18}O$ and δ^2H) and other parameters (DO saturation, pH, Cl^-, Ca^{2+}, NO_3^- and PO_4^{3-}) were collected in late May, mid-August, and late October 2016.

For analysis purposes, the Jordan River was divided into nine segments between Utah Lake and Redwood Road. Most river segments were defined at breaks within 1 kilometer (km) above or below a WRF (Figure 2-9). Other segments were defined at Willow Park (about 5 km below Utah Lake) and at the 2100 South diversion. The most downstream segment (segment 9) extends from the diversion downstream to Redwood Road.

A Bayesian Simple Linear Mixing (SLM) model was used to estimate the contribution of different water sources (i.e., Utah Lake, groundwater, tributaries, effluent, and return flow from canals) to each study segment of the Jordan River. Source values of $\delta^{18}O$ and δ^2H from measured flow sources were used in the model as well as values from literature for groundwater and tributaries.

Results from the SLM model were compared to hydrologic mass balance (HMB) calculations to assess the certainty of water source contributions to the river. HMB data was collected from water reclamation facility (WRF) operators, and gaging stations (federal, state, and county agencies). Return flows were determined as the difference between measured upstream and downstream discharge. Comparison of results between the two models indicated the two approaches had ≤10 percent difference between estimates for each source for 57 percent of all data points and ≤20 difference for 83 percent of all data points. On average, estimates differed by 11 percent in the spring and 10 percent in the summer. Fall comparisons were not completed due to equipment failure.
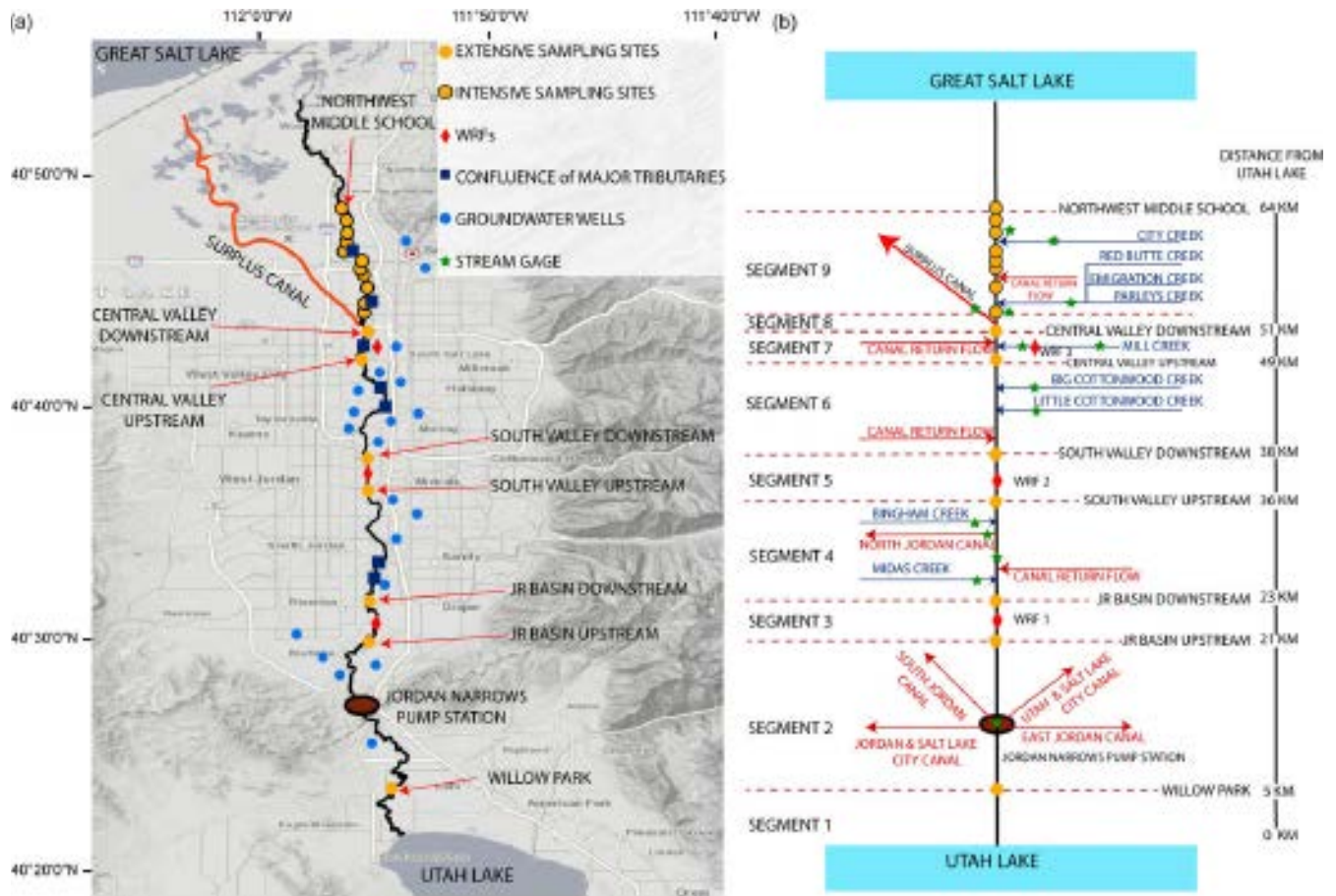
Figure 2-9. Jordan River (black line) flowing through the Salt Lake valley and connecting Utah Lake to the Great Salt Lake. Surplus Canal diverting water from Jordan River to the Great Salt Lake is shown as a red line. **[Blue circles are the location of groundwater wells to inform groundwater end-member in this study. Blue squares represent the confluence of different tributaries and the Jordan River. Red diamonds are the location of the different water reclamation facilities (WRFs). Orange circles are sampling sites (intensive sites have a black border). Jordan Narrows pump station (brown oval) diverts water from Jordan River to different irrigation canals. (b) Schematic representing nine river segments and source water inputs and outputs within each segment. This schematic was used to inform data parameterization of our Bayesian simple linear mixing (SLM) model. Source: Follstad Shah et.al. (2019a) Figure 2.]**

Results indicated that dominant water sources to the Jordan River varied spatially and temporally over an annual cycle (Figure 2-10). The most substantial change in flow occurred in the spring season below the addition of tributary discharge and flow diversions to the Surplus Canal. During the spring season, Utah Lake was the largest flow contributor (>65 percent) above river km 21 (upstream of Jordan Basin Water Reclamation Facility [JBWRF]), then declined to 25–38 percent of flow between river km 23–36 (above SVWRF), and was ultimately the lowest fraction of flow (3–12 percent) after river km 38 (below SVWRF). Groundwater represented 21–35 percent of flow through river km 21 (upstream of JBWRF). Below this point, groundwater was augmented by flow from tributaries.
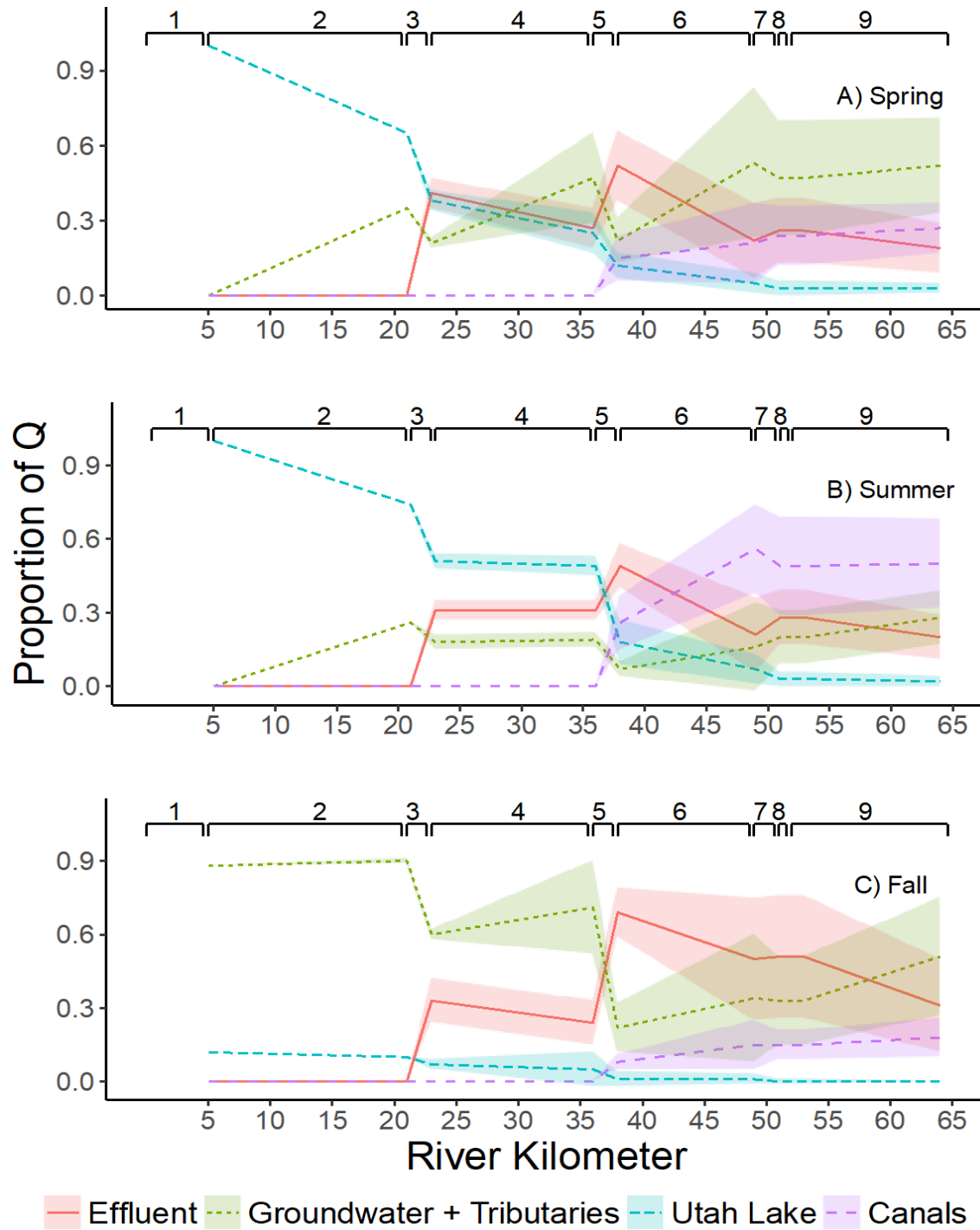
Figure 2-10. Proportion of river water attributed to different water sources along the Jordan River during three seasons based on estimates from a Bayesian SLM model using water isotopes of $\delta^{18}O$ and δ^2H for each source. Shading indicates the standard deviation of the estimated proportion for each water source within individual river segments. Groundwater and tributary sources are combined because their isotopic signatures were indistinguishable. Brackets indicate river Segments 1-9. Source: Follstad Shah et al. (2019a) Figure 4.

During the summer, water from Utah Lake was again the dominant contributor of flow (49–74 percent) upstream of km 36 (above SVWRF) but was surpassed by effluent inputs (49 percent) below river km 38. Effluent contributions quickly declined to 20–28 percent of flow downstream, with increased canal inputs that contributed half of total flows from river km 49 (above CVWRF) downstream to river km 64 (near Redwood Road). During the summer, groundwater contributed less flow (≤26 percent) upstream of river km 21 compared to spring. Downstream of river km 21, combined groundwater and tributary inputs provided ≤25 percent of flow.

During the fall season, discharge from Utah Lake dropped to 5–12 percent of flow upstream of river km 38, in contrast to much higher contributions in the spring and summer seasons. Combined groundwater and tributary inputs became the greatest contributor to flow (>60 percent) through river km 36, as well as downstream of flow diversions to the Surplus Canal (51 percent). However, no tributary inputs occurred prior to river km 23, while they exceeded groundwater inputs downstream of river km 53 (diversion to the Surplus Canal). A shift toward groundwater as the primary water source above river km 21 occurred in the fall. Less flow from Utah Lake and tributaries in fall relative to other seasons allowed effluent to dominate flow in river km 38–53 (50–69 percent). Canals contributed ≤25 percent of flow in river km 49–64 during the spring and fall seasons.

Water quality measurements varied temporally in similar patterns to those documented in the Phase 1 TMDL. Some of the spatial variations observed can be explained in part using flow-source information. Spatial variability in DO saturation was greatest in summer, showing sharp decreases downstream of the Surplus Canal. DO saturation also declined by 30 percent from summer to fall below this point, which coincided with a decrease in Utah Lake contributions (64 percent) and a concomitant increase in groundwater. DO saturation also increased in all seasons in segments receiving tributary discharge. Elevated water temperatures in summer were attributed to inputs from effluent, canals, and tributaries. Effluent discharge is considered the primary source of nutrients given their concentrations are an order of magnitude greater than the river and they are a major source of flow in the LJR during the summer and fall season.

Mass balance model results based on isotope tracing indicate that sources of primary flow were substantially different above and below river km 38 (downstream of SVWRF) in all seasons. Dominant sources of flow also varied by river segment throughout the year. Measured flow from the three WRFs discharging to the study reach remained consistent in the spring and fall at about 3.4 m^3/sec. However, based on changing inputs from other sources, total percent contribution from effluent was 19 percent in spring and 63 percent in fall. Furthermore, source-water inputs at the beginning of the survey reach (e.g., Utah Lake and groundwater) were evident in the LJR despite having low inputs initially compared to other sources.

Shifting from water sources to flows, earlier studies examined the use of flow manipulation as a potential mechanism for increasing DO during late summer when the LJR is experiencing chronic low DO conditions (SWCA 2014). Acquiring sufficient water rights has slowed implementation of an actual flow experiment. In the interim, analysis of existing instantaneous flow and diel DO collected 2011–2016 from four permanent sensor locations on the LJR indicated the potential for increasing DO in response to dry weather (non-storm) flow increases (Cirrus 2016).

Representative flow changes in dry weather periods at the 2100 South diversion were identified and paired with available DO data from each downstream LJR monitoring site. The DO data record was adjusted to account for travel time between the 2100 South station and each DO sonde using the Salt Lake County HEC-RAS model. After flow-DO data sets were paired, mean DO was calculated for 24 and 48 hours before each flow change as well as 24 (Period 1), 48 (Period 2), and 72 (Period 3) hours after a flow change at each monitoring site.

A total of 45 events were identified in the analysis after screening and removing some data points due to intermittent or questionable measurements and measurable rain events that occurred in the period of analysis following the flow change. The average flow change at the 2100 South diversion was an increase

of approximately 30 cfs, ranging from a 65 cfs increase to a 26.2 cfs decrease. Most flow change events were increases within a narrow range of 25–35 cfs.

The precision of the sondes from which the DO data was derived was assumed to be +/- 0.2 mg/L. In terms of data analysis and interpretation, the range of variables affecting DO in the LJR, coupled with limitations on the quality and quantity of data available for review, made establishing a causal relationship between increased flow and DO concentration impossible. Focus was directed at DO changes correlated with rather than caused by flow changes. Beyond that, limited data availability, and thus small N sizes, precluded any meaningful statistical comparison of mean differences among DO changes.

Results were mixed over the 45 flow-change events reviewed at the four LJR locations. In Period 1, 22 flow-increase events were associated with a decrease in DO, while remaining events indicated an increase. Downstream segments in the LJR (DWQ segments 1 and 2, downstream of North Temple) had 64 percent (16) of the events with a negative DO response to a flow change compared to upstream segments in the LJR (DWQ segment 3, upstream of North Temple). The monitoring station at 300 North had the greatest net negative response to flow increases, where six of the nine events were negative.

In general, DO rebounded during Period 2 at three of the four monitoring locations, where 24 events were associated with an increase in DO. The only site showing a continued decline in DO was 300 North. Period 3 DO for three stations moved toward pre-flow-change values. Period 3 DO at 300 North continued to steadily decline but DO levels at the three other stations remained above pre-flow-change concentrations.

Results suggest a positive DO change from flow increases without 300 North, comparable to summer season results. The averaged net changes in DO across all measured events and sites were 0.04 mg/L (Period 1), 0.13 mg/L (Period 2), and -0.01 mg/L (Period 3).

To validly interpret these results, considerations should be made for the small mean differences in DO before and following flow changes, the narrow (i.e., 25–35 cfs) range of flow changes available for the study period, and the seasonal differences observed. As noted above, the limited data and flow range would not support a rigorous statistical analysis.

Separating the flow changes by season identified two flow-change events (July and August 2015) measured at three sites during the 2015 summer season. Both events showed a positive change in average DO at 2100 South, Cudahy, and Burnham Dam, for all periods measured including a river-wide average increase of 0.15 mg/L (Period 1), 0.36 mg/L (Period 2), and 0.32 mg/L (Period 3).These gains are important given the fact that the Cudahy and Burnham sites typically see more DO violations on the river than any other sites monitored (Table 2-1). However, spring events showed mixed results in terms of DO response.

In conclusion, flow could play a role in reducing oxygen demand and potential DO violations. However, positive DO responses could be delayed, and the potential to depress DO levels in the LJR in response to flow change during critical times may exist. The two summer-season flow changes showed a gain at the 2100 South, Cudahy, and Burnham sites and a similar response at different flows. This increase and consistency could indicate positive benefits in response to increasing summertime flows.

2.2 DO DEMAND IN JORDAN RIVER SEDIMENTS

The Phase 1 TMDL characterized DO demand in the river channel sediment due to aerobic decomposition of OM in sediments and oxidation of inorganic compounds. Together, these processes combine to produce SOD. SOD measurements at the time were collected from seven locations (Goel and Hogsett 2009, Goel and Hogsett 2010) and ranged 0.84–3.37 g/m^2-day. SOD measurements generally increased with distance downstream from 2100 South. Winter measurements were higher than summer measurements at some locations. Average SOD for all seasons and unadjusted for temperature was 1.7 g/m^2-day. The final water quality model (QUAL2Kw) was calibrated using measured DO levels in the LJR to prescribe SOD levels of 1–3.5 g/m^2-day.

Table 2-1. Averaged Net Change in DO (mg/L) across measured events at 24 (Period 1), 48 (Period 2), and 72 hours (Period 3) after flow change.				
	N	Period 1	Period 2	Period 3
2100 South	15, 13, 12	-0.04	0.13	0.06
300 North	8, 8, 7	-0.12	-0.14	-0.33
Cudahy	6, 6, 5	0.19	0.29	0.08
Burnham	12, 10, 9	0.03	0.20	0.12
River Average	**45, 41, 37**	**0.04**	**0.13**	**-0.01**
Averaged net change during two summer flow events only (July 17 and August 12, 2015)				
2100 South	2	0.05	0.35	0.32
Cudahy	2	0.15	0.32	0.24
Burnham	2	0.26	0.41	0.40
River Average		**0.15**	**0.36**	**0.32**
Source: Cirrus 2016 Table 3–4.				

The 2017 synthesis reviewed a substantial number of SOD measurements and analysis (Hogsett 2015). These results indicated SOD had a significant influence on oxygen levels in the LJR, accounting for over 50 percent of the ambient DO deficit during 84 percent of the sampling events (N = 46) and over 75 percent of the DO deficit during 58 percent of the sampling events (N = 32). Based on seasonal average SOD values of 2, 1.8, and 1.5 g O_2/m^2-d for LJR segments, SOD accounts for 54 percent of the ambient DO deficit in the LJR (Hogsett 2015).

Other important observations from the previous synthesis describing DO demand in Jordan River sediments include:

- Values increased during the winter, contrary to what was expected due to cold winter temperatures. This could be due to groundwater upwelling, greater periphyton growth and respiration in response to improved water clarity, decomposition of seasonal OM loads, microbial activity tolerant to environmental change, or diffusion of reduced chemicals that limit SOD and are not affected by temperature.

- SOD consists of methane oxidation and aerobic metabolism. Methane production from sediments contributes 56 percent of total SOD.

- The life-cycle of sediment OM indicates that any surface OM under aerobic conditions is likely removed within a year of deposition, particularly if channel sediments are not disturbed. The life-cycle of sediment OM under anaerobic conditions ranges from 2.5 to 4.8 years. Due to this extended life cycle and incoming annual loads, OM is steadily accumulating in the LJR under anaerobic conditions.

- Based on DO monitoring during dredging, oxygen demand can be enormous if sediment is disturbed at depth but insignificant at shallow depths on small scales. Based on these assumptions and under anaerobic conditions, the life cycle of OM is limited by oxygen transfer rates and could potentially be shortened if sediments were disturbed.

There is currently little information defining the influence of nutrients on SOD and rate of OM decomposition in the sediments. Other areas where additional understanding is needed include nutrient

transformation in sediments, influence of sediment temperature on SOD and sediment microbial activity, and spatial distribution of SOD in the LJR. The remainder of this section will review the results of recent research that are applicable to these data needs.

2.2.1 Sediment Oxygen Demand and Nutrient Processes

Decomposing OM in the sediment creates a demand for DO in the overlaying water column. Decomposition processes in sediment are influenced by nutrient dynamics in sediment. Both processes are important to understand in evaluating an oxygen budget.

SOD measurements were collected from two sites on the LJR including 1300 South and the Legacy Nature Preserve in July and September 2015 (Goel and Abedin 2016). Benthic nutrient flux measurements were collected at the same sites from two sediment chambers and two water-column chambers to measure daytime nutrient dynamics at the sediment-water interface. The first four hours of nutrient flux measurements were collected under ambient conditions. The remaining four hours of the experiment were measured after chambers were spiked with nutrients to determine sediment reaction to a nutrient pulse. Potential denitrification and nitrification for each site were calculated using lab measurements of sediment slurry collected during field surveys. Ammonia oxidizers and denitrifiers were also identified at each site using biomarkers and quantitative polymerase chain reaction analysis to verify the presence of denitrifying genes and quantify the number of functional genes present at each site, respectively.

SOD measurements ranged from 2.4 to 2.9 g-DO/m^2-d at the two Jordan River sites. These rates are comparable to previous measurements collected from the LJR (Hogsett 2015). Percent of ambient DO deficit associated with SOD for each sampling event ranged 72–97 percent at 1300 South and 72–90 percent at the Legacy Nature Preserve. Higher values measured in September were considered to be due to increased OM input from falling leaves and the resulting increase in bacteria metabolism. Based on SOD$_{T25C}$ values (Butts and Evans 1975, Hogsett and Goel 2013), benthic sediment conditions at 1300 South are moderately polluted while Legacy Nature Preserve sediments are considered moderately polluted to polluted.

Positive and negative nutrient flux measurements can be a response to several biogeochemical processes at the water-sediment interface. Nutrient flux measurements indicated that both sites are a sink (i.e., negative flux) for nutrients, especially regarding nitrate (Figure 2-11). Simultaneous negative fluxes of N (ammonia and nitrate) and P may be caused by algal uptake. This response was observed at 1300 South under ambient (unspiked) conditions. At the Legacy Nature Preserve site, nitrate flux was negative under all conditions. When the nutrient spike was introduced, P flux also became negative, which could be the result of higher algal uptake due to bioavailable nutrients.

Nitrification rates varied 0.008–0.07 mg-N/g-day. Comparing nitrification rates with flux measurements suggests dominance of denitrification over nitrification processes in the LJR. Measured rates of nitrification and denitrification identified an increase in both processes when bioavailable nutrients were added to measurement chambers. DNA analysis of sediment samples identified numbers of denitrifying and nitrifying genes at study sites similar to those of engineered ecosystems (i.e., wastewater treatment) which subsequently indicate a high potential for N removal and ammonia oxidation in the LJR under favorable conditions of nutrients, O, and temperature.

Stable isotopes were also used to identify N transformation in the Jordan River (Follstad Shah et al. 2017). Seasonal analysis of measurements in the LJR indicated N transformation via denitrification during the spring season only, based on correlation between δ^{15}N-NO$_3$ and δ^{18}O-NO3 (Figure 2-12). Longitudinal analysis from all seasons indicated samples were less enriched in δ^{15}N-NO$_3$ with distance downstream, indicating N fixation or contributions from less enriched sources (e.g., leaf litter from N$_2$-fixing species such as Russian olive, groundwater recharge, etc.). The authors suggest these results indicate that denitrification does not have a strong impact on the Jordan River and that nitrification may be favored. This conclusion is speculative because the authors did not consider nitrate inputs with varying isotopic composition, and the conclusion runs counter to sediment N fluxes measured by Goel and Abedin (2016).
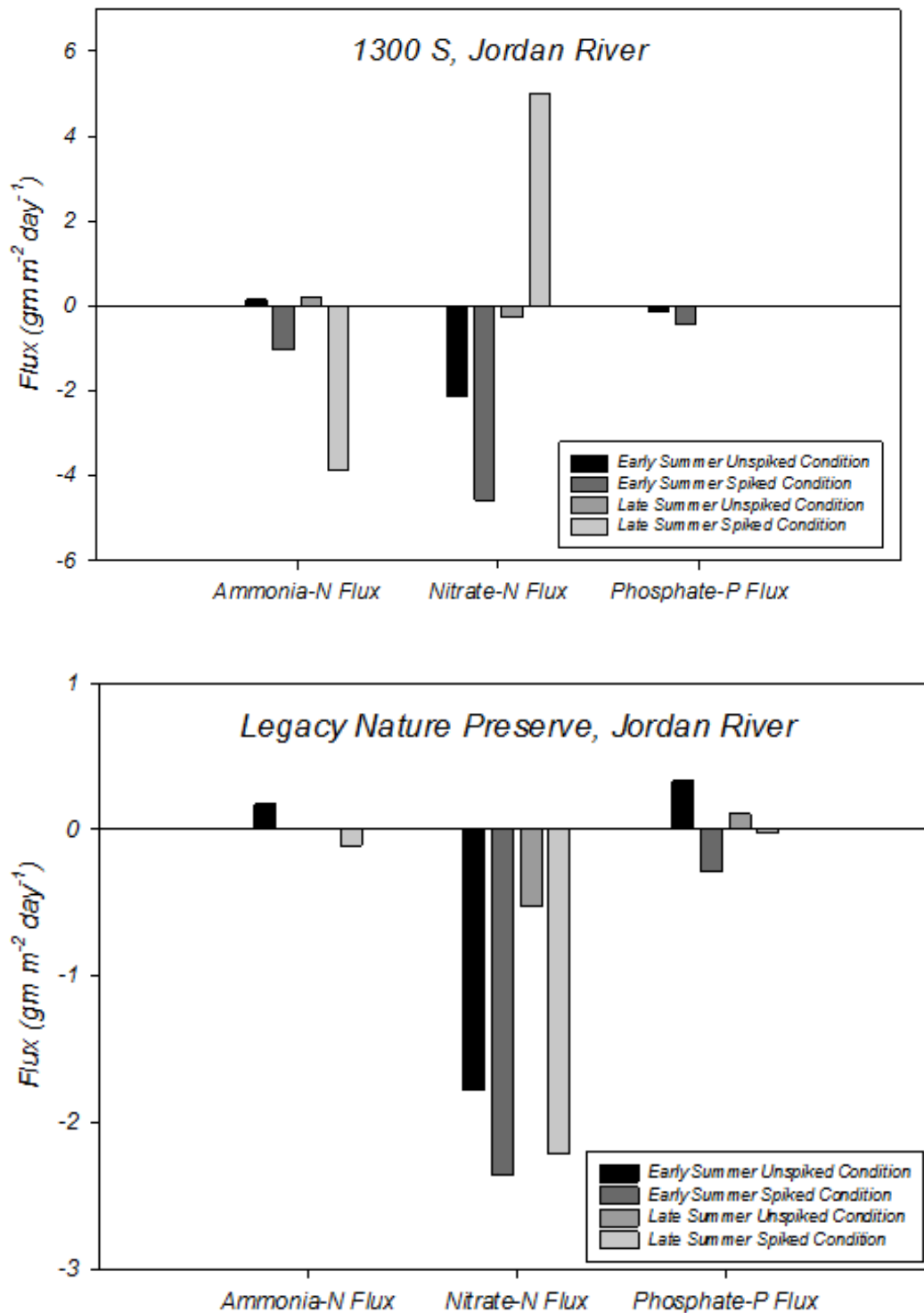
Figure 2-11. Measurements of ammonia, nitrate, and phosphate flux from lower Jordan River sites under unspiked and spiked conditions during early (July) and late (September) summer 2015. Source: Goel and Abedin (2016) Figure 6.
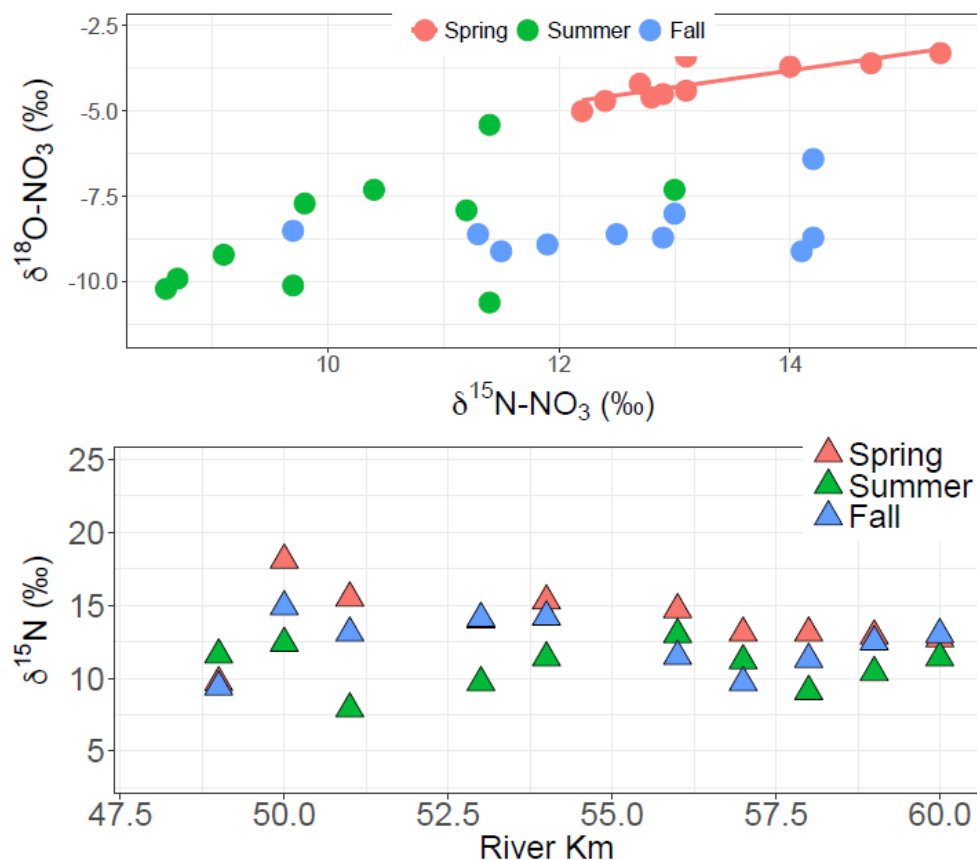
Figure 2-12. Seasonal analysis of δ^{15}N-NO$_3$, δ^{18}O-NO3, and δ^{15}N(‰) for 10 study sites in the LJR during measured during spring, summer, and fall 2016. Upper graph indicates dual enrichment of δ^{15}N-NO$_3$ and δ^{18}O-NO3 during the spring (r2=0.45, p<0.05) which theoretically signifies potential removal from the water column as denitrifying microbes preferentially convert isotopically lighter forms of NO$_3^-$ to N$_2$0. Although this trend is apparent in spring, it is not during the summer or fall seasons. For this relationship to signify NO$_3^-$ removal, dual enrichment would need to occur in the downstream direction. The lower graph shows a decreasing trend of δ^{15}N-NO$_3$ and a downstream net NO$_3^-$ production in the LJR. Source: Follstad Shah et al. (2017) Figure 6.

Microbial activity in sediments can be determined by examining ecoenzymes associated with acquisition of C, N, and P (Follstad Shah 2017). Sediment samples were collected from the Jordan River during May, August, and October 2016 and measured for EEA including BG, NAG + LAP, and AP. Sediment ecoenzyme activities showed a high level of spatial and temporal variability (Figure 2-13). Spatial variability showed similar longitudinal patterns of EEA compared to measurements collected from the water column (discussed above in Section 2.1.2). Consistent positive relationships were also identified between BG vs. NAG+LAP, BG vs. AP, and NAG+LAP vs. AP that explained 11–51 percent of the variability. Slopes for C relationships were similar to or >1, indicating greater allocation of energy to acquiring C. Slopes of NAG+LAP vs. AP were roughly 1 during the spring and fall seasons and 0.74 in summer which indicated a greater energy allocation to acquiring P during the summer when growth rates are highest.
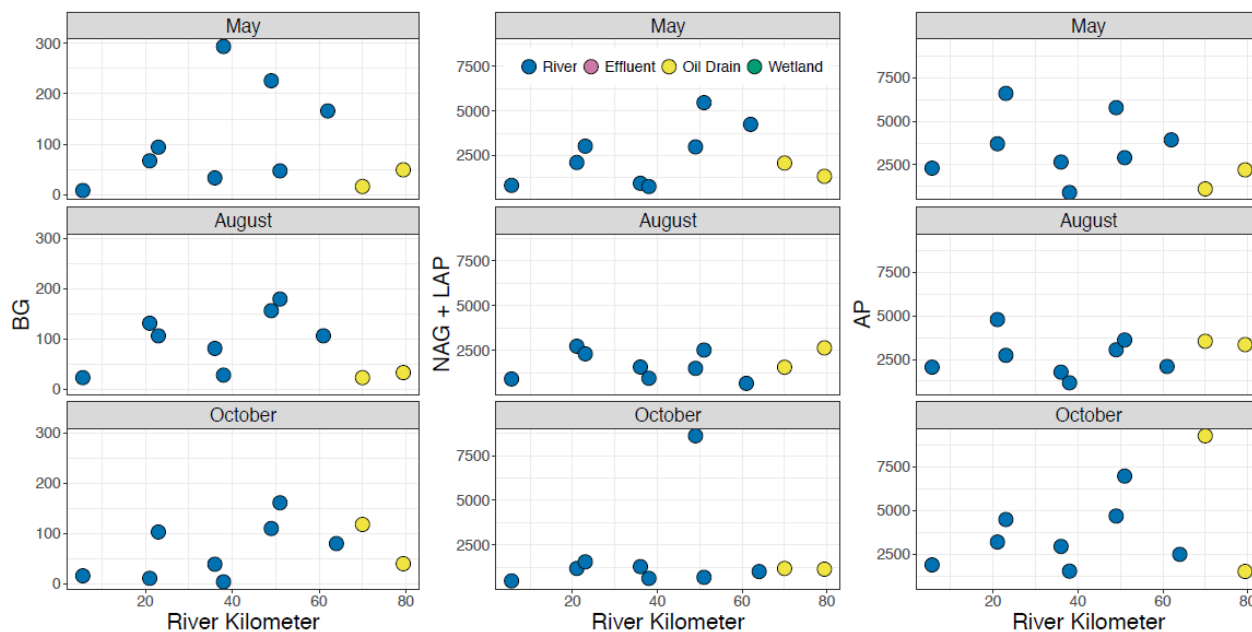
Figure 2-13. Ecoenzyme activities (nmol/L-h) associated with acquisition of C (BG), N(NAG + LAP) and P (AP) are variable both spatially and temporally in sediment of the Jordan River (blue circles), oil drain (yellow circles), and wetland (green circles). Source: Follstad Shah (2017) Figure 13.

Based on these results, microbial activity in the sediment is limited by C during some seasons as well as by P at certain times of the year. Microbial communities in the water column and sediment appear to have an ample supply of N throughout the year to support growth.

2.2.2 Macroinvertebrate Influence on SOD

Measurements of SOD reviewed in the previous synthesis and in this synthesis account for the portion of oxygen demand created by any macroinvertebrate processes including the influence of *Corbicula* and *Potamopyrgus*. A recent literature review identified characteristics of these species and bivalves similar to *Corbicula*, as well as oxygen demand processes in Jordan River sediments to which these species may contribute. The review extrapolated potential impacts on water quality and sediment processes in the Jordan River based on reported literature values and estimates of population densities. This section summarizes those results as reported in Richards (2018).

Based on literature values and average estimated densities of *Corbicula*, as much as 1 kg C/m2 could be sequestered in shell production rates. These rates could isolate about 10 metric tons C/km-yr in some segments of the Jordan River (Richards 2018). Note that this process is sequestering inorganic C and not OM. Other benefits include a consumption (based on *Corbicula* density 1,436/m^2) of N (mean = 1.56 mg/m^2-day) and P (mean = 0.56 mg/m^2-d). Consumption of sediment OM is discussed below in Section 3.

Negative impacts on Jordan River water quality produced by *Corbicula* from production/excretion of nutrients were reported as 165 umol/m^2-day for ammonia 33 umol/m^2-day for P (Richards 2018). However, these rates are questionable as they indicate that *Corbicula* excrete roughly twice as much nutrients as they assimilate. Mean oxygen consumption and CO_2 respiration rates in run habitats of the Jordan River are 2.10 mg/m^2-hr and 1.0 mg/m^2-hr, respectively (Table 2-2). Total oxygen demand would also account for other losses that would increase the net loss of oxygen. The ability of *Corbicula* to improve low oxygen

conditions is dependent on reducing organic detritus contributions to sediment that results in oxygen demand through heterotrophic respiration and decomposition of OM.

Assuming that estimated densities are correct, *Potamopyrgus* densities are projected to have a production rate of about 1,500 mg AFDM/m^2-day, excretion of approximately 8 mg N/m^2-day, and egestion of 200 mg N/m^2-day (Richards 2018).

Table 2-2. Estimated O$_2$ consumption and CO$_2$ respiration rates (mg/m^2-hr) by *Corbicula* in run habitat sections of the Jordan River downstream of CVWRF to 900 South.				
	Corbicula Density (m^{-2})[a]	*Corbicula* Dry Weight (g m^{-2})[b]	O$_2$ consumption (g/m^2-hr)[b]	CO$_2$ respiration (mg/m^2-hr)[c]
Median	650	0.52	0.001	0.0009
Mean (± SE)	1436 (910, 1962)	1.15 (0.73, 1.57)	0.002 (0.0013, 0.0027)	0.0017 (0.0011, 0.0023)
75th percentile	1,223	0.98	0.0017	0.0014
95th percentile	3,700	2.96	0.0049	0.0041
99th percentile	12,400	9.92	0.0159	0.0135

[a] Jordan River *Corbicula* density estimates downstream of CVWRF in non-pools (Richards 2017, see table 59).

[b] Based on Hakenkamp and Palmer (1999) *Corbicula* dry weight estimates and regression model: oxygen consumed = 0.19 + (1.58 x *Corbicula* dry weight (g)).

[c] Based on Bott (2007) Respiratory Quotient: 1 mol CO2 respired/1 mol O2 consumed = 0.85

Source: Richards 2018 Table 63. Note that rates of consumption and respiration have been converted to g/m^2 – hr for comparison purposes to SOD measurements.

3.0   TOTAL ORGANIC MATTER AND ORGANIC MATTER SOURCES

This section summarizes relevant findings of the Phase 1 TMDL and the 2017 synthesis. Sections 3.1–3.6 go on to discuss the results of recent research (2015 to present) that build on our previous understanding of OM dynamics in the LJR.

Organic matter in the Jordan River comes from a combination of sources located away from the channel as well as sources that create OM in the water column and sediments of the channel itself. Natural sources of OM outside of the channel are described as terrestrial sources, and OM produced within the channel is defined as an aquatic source.

The Phase 1 TMDL identified OM as the pollutant of concern leading to low levels of DO in Jordan River Segments 1–3 (DWQ 2013). Sources of OM were classified into size categories including coarse particulate organic matter (CPOM), fine particulate organic matter (FPOM), and DOM (Figure 1-1). In most research, DOM and DOC are used interchangeably. This synthesis will use DOM when discussing OM measuring less than 0.45 um, due to how this particle size was reported in the reviewed research.

Each size class can be delivered to segments of the upper or lower Jordan River by one or more OM pollutant sources, and each source can contribute to more than one size category. Details describing particle size, transport, and fate of OM loads and sources are important only to the degree they help to define a link between a pollutant source and a water quality process that generates oxygen demand in the LJR.

The Phase 1 TMDL defined TOM in the Jordan River as a combination of living and dead material in a range of coarse and fine particle sizes. Little or no information was available to quantify how TOM loads change as they move through upper segments and into the LJR or how OM loads vary among seasons and years.

The exact proportions of OM size categories were not known in the Phase 1 TMDL. OM particle size categories provided useful information in support of critical processes that affect movement and deposition of OM between sources and the LJR. TOM loads were defined with a combination of direct measurements and computer modeling. These loads were assumed to include all particle sizes and OM loads that could influence DO in the LJR.

The 2017 synthesis identified several important pieces of information regarding OM missing in the Phase 1 TMDL including:

- OM budget results identified DOM as the dominant particle size. Loads of CPOM were typically less than 5 percent of the TOM load in transport. Two methods were used to measure CPOM during the 2013 water year, one to define OM budget results and the other as part of long-term monitoring. Results of both studies showed large differences and were noted to have a high amount of variability between monitoring dates (Epstein et al. 2014, Miller 2015). The synthesis noted that additional documentation was needed to allow DWQ to resolve discrepancies, and that a review of raw data may be necessary.

- Based on the magnitude of flow observed during spring runoff and autumn rain events, the total annual CPOM load delivered to and carried by the Jordan River was estimated at 200,000 kg/year during normal to high runoff years.

- Monthly monitoring identified the highest concentrations of DOM during the fall season, corresponding to leaf drop. River segments with the largest OM pool included Upper Jordan River (UJR) segments 7800 South–5400 South and 3300 South–2300 South due to unmeasured inputs of FPOM and DOC.

- UJR segments were observed to have high levels of primary production in the water column and benthos. Results indicated that autotrophic net daily metabolism in UJR segments could contribute 55 percent of the OM load estimate used in the Phase 1 TMDL for this reach.

- Isotope mixing model results showed DOM was derived primarily from terrestrial sources, FPOM was comprised of equal proportions of terrestrial and aquatic sources, and CPOM was primarily from terrestrial sources. Terrestrial sources include OM generated outside of the stream channel by natural and wastewater sources. Aquatic sources include OM generated inside the stream channel (e.g., phytoplankton, benthic algae, etc.), C:N ratios of OM more clearly indicated that FPOM was similar to aquatic sources.

- Episodic releases of highly degradable DOC from storm events flushing the stormwater conveyance system could be related to DO depletion events in the LJR. The remaining particulate OM in stormwater could settle to the channel bottom and contribute to SOD. The deposition of residual, more recalcitrant particulate OM from stormwater flushing and fall leaf drop could contribute OM to SOD, but the year-round SOD is likely related to a variety of OM sources.

The 2017 synthesis did not include information on the nature of OM (labile versus recalcitrant). Although isotope tracing results provided some information regarding OM sources (i.e., terrestrial versus aquatic), additional details that more clearly link pollutant sources to OM types are needed to support load allocations in the Phase 2 TMDL. Seasonal analysis of OM loading and projected future trends that influence loading will also be required in the upcoming TMDL. Finally, information that identifies opportunities and cost-effective strategies for improving DO levels will benefit the TMDL process by providing options for meeting load allocations and water quality endpoints.

3.1 COARSE PARTICULATE ORGANIC MATTER

Measurements of CPOM were collected during most months 2010–2014 from major tributaries and at selected locations along the Jordan River (Miller 2019c). Tributary measurements were collected above and below debris basins near the Salt Lake Valley margin and at the Jordan River confluence. Standard sampling protocols were not available when sample collection began. When flow conditions were <300 cfs, width and depth integrated samples were collected using a standard 10 x 18-inch sweep net. During high flows (>300 cfs), samples were pumped through a 3-inch hose from the center of the channel at the bottom of the water column. The intake end of the hose was weighted with approximately 40 pounds of steel plate to ensure the pump screen remained on the stream bottom. Flow measurements used for load calculations came from available gage data or were measured using a Valeport mechanical flow meter.

Peak monthly CPOM loads from tributaries typically occurred May–June of each year and were influenced by flow rate and location. Annual precipitation levels in 2011 were approximately 30 percent above average, and tributary loads were generally higher compared to other years. Peak monthly loads were measured from Mill Creek and City Creek October–December of some years despite low flows, indicating an anthropogenic source of OM. Measurements of CPOM showed that debris basins were effective (≥ 70 percent) at removing substantial loads from tributary flow. Results included some high flow events where little or no removal occurred. On those dates it is assumed the basins had not been cleaned out or that high flows provided insufficient residence time for settling to occur.

Substantial loads were contributed to Jordan River tributaries in reach segments between debris basins and the confluence of each stream with the Jordan River. Peak monthly loads from tributaries below debris basins were reported in 2011 for Little Cottonwood Creek and Big Cottonwood Creek at 42,000 kg and 75,000 kg, respectively (Miller 2019c).

CPOM loads calculated at LJR sites allow the estimation of the amount of OM lost to deposition between site locations. Figure 3-1a indicates differences between monthly loads during 2011 at 1700 South and 300 North, suggesting some deposition in this segment. However, a review of loads in 2014 (Figure 3-1b)

indicates that loading at 1700 South is lower than 300 North for several months including May, June, August, September, November, and December.

Due to flow management at the 2100 South diversion, peak monthly loads at different times between years and sometimes multiple peaks within a given year. Monthly loads at 1700 South and 300 North show some increase in loads during May and June and additional peaks during summer and fall. These loading patterns reflect the combined influence of the 2100 South diversion, tributary inflow, and stormwater discharge during different times of the year.

Measurements of Jordan River OM loads collected in 2013 by Epstein et al. (2016) were compared to CPOM measurements collected by Miller (2019c; Table 3-1). Miller's CPOM measurements are generally 2–5 times larger than those reported by Epstein et al. (2016). Other OM particle size measurements (i.e., FPOM and DOM) were not collected by Miller, so further comparison to identify consistent differences in OM particle sizes between the two studies is not possible. As stated in the 2017 synthesis, measurements of CPOM samples collected during both studies were noted to have a high amount of variability between monitoring dates (Epstein et al. 2016, Miller 2015). Consequently, there is potential for high variability among samples, sampling methods, and sampling events. Review of original data records and further analysis could provide insight into why some of these differences may exist.

Table 3-1. Comparison of CPOM measurements collected during a wet year (2011) and a dry year (2013) by Miller (2019c) and during 2013 by Epstein et al. (2014).

Site	2011 - WFWQC		2013 - WFWQC		2013 - Epstein and Baker
	AFDM (kg)	C (kg)	AFDM (kg)	C (kg)	C (kg)
1700 South	218,712	72,175	33,954	11,205	6,023
300 North	62,119	20,499	139,433	46,013	8,592[a]
Legacy Nature Preserve	74,836	24,696	121,547	40,110	14,373[b]

[a] Sample collected at 500 North.
[b] Sample collected at Cudahy Lane.

Source: Miller (2019c) Table 4.
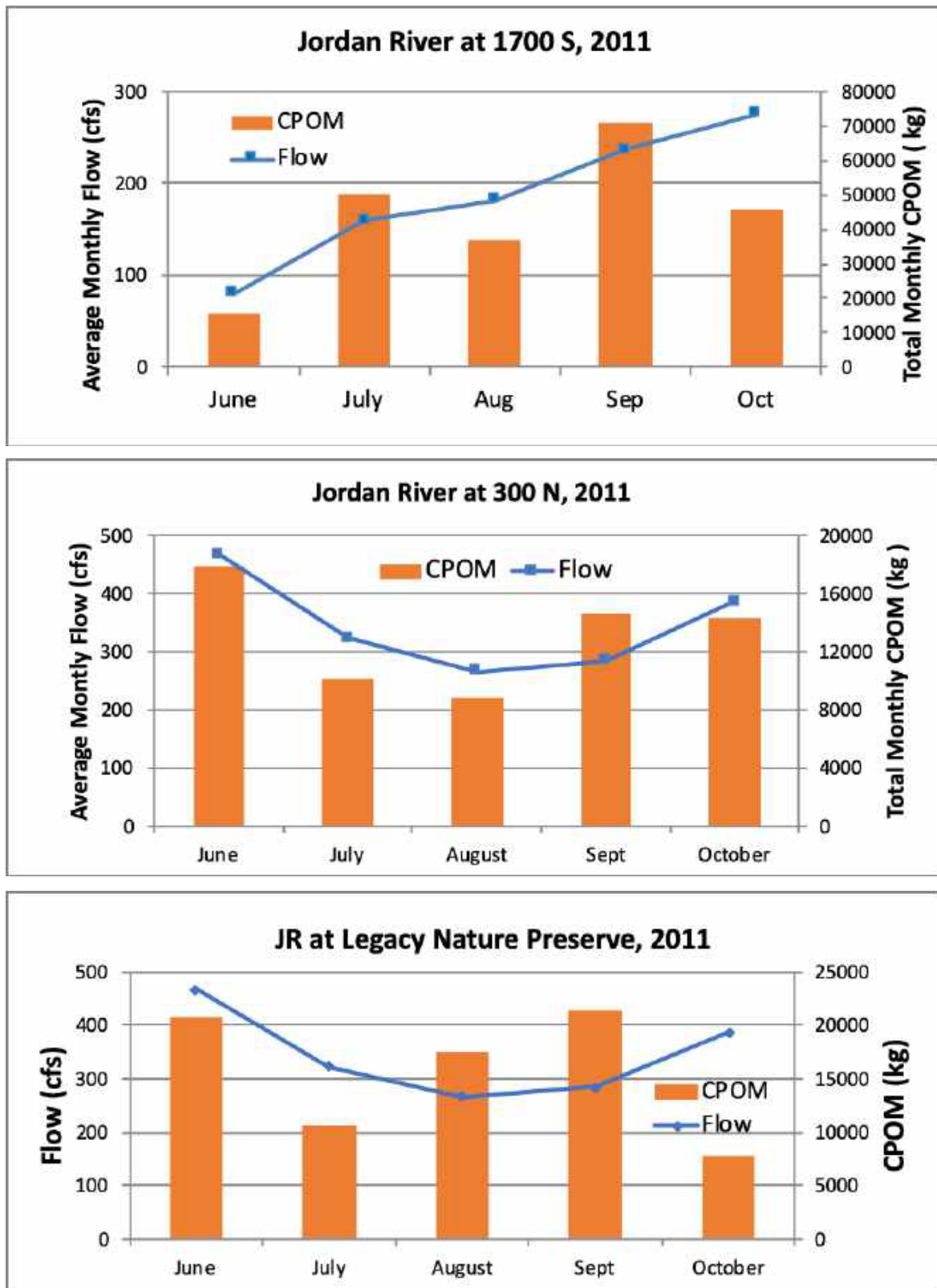
Figure 3-1a. Measurements of flow and CPOM during a wet year (2011) from LJR monitoring sites located at 1700 South, 300 North, and Legacy Nature Preserve. Flow values are from stream gages located at 1700 South and 500 North. Source: Miller (2019c) Figure 54.
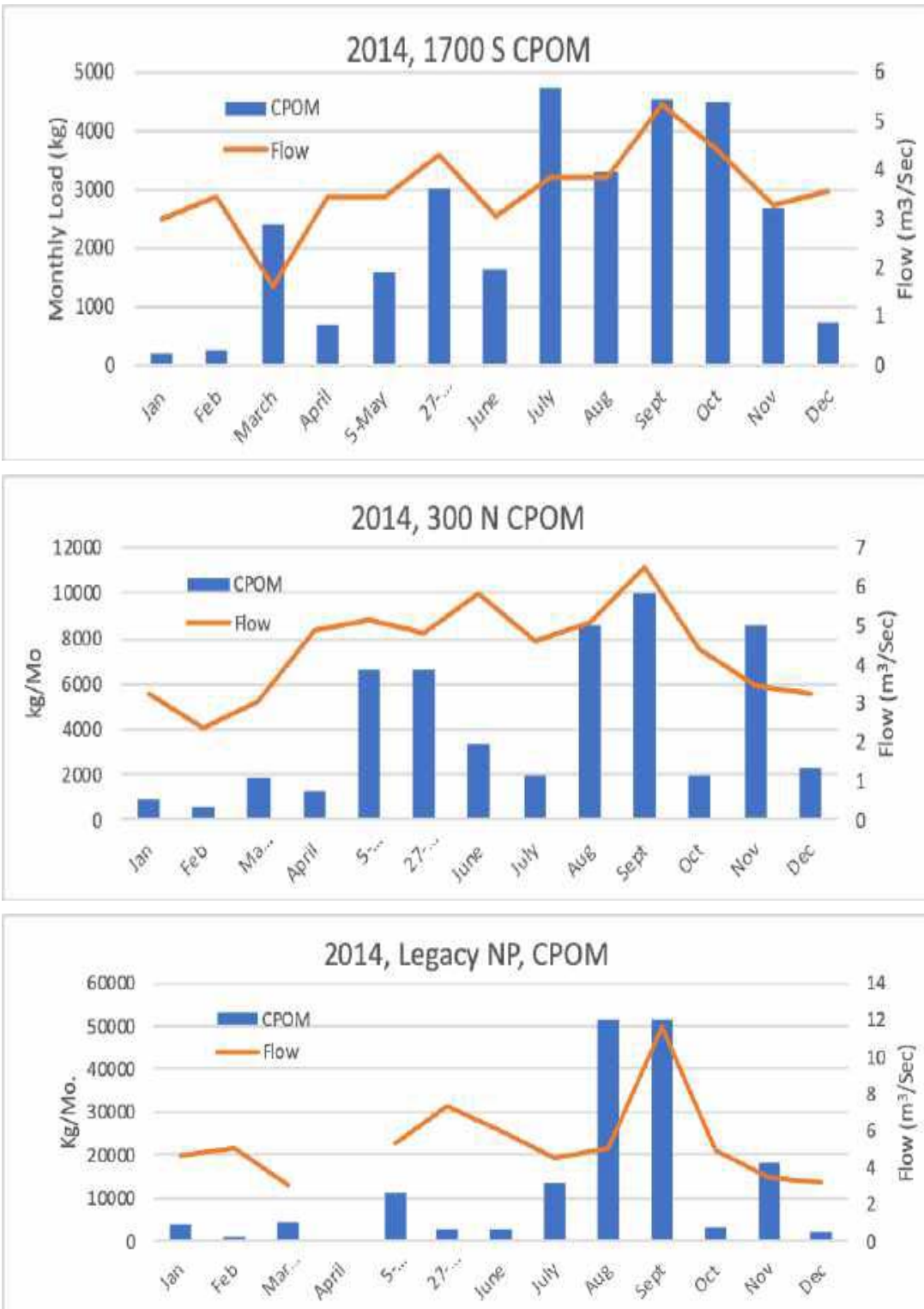
Figure 3-1b. Measurements of flow and CPOM during a dry year (2014) from LJR monitoring sites located at 1700 South, 300 North, and Legacy Nature Preserve. Flow values are from stream gages located at 1700 South and 500 North. Source: Miller (2019c) Figure 57.

3.2 FINE PARTICULATE AND DISSOLVED ORGANIC MATTER

Particles >1mm in diameter are classified as CPOM. FPOM is considered to include organic particles < 1 mm and >0.45 mm in diameter. DOM is OM that passes a nominal 0.45-µm filter. Although measurements of volatile suspended solids (VSS) are generally considered to represent FPOM, the methods and design of grab sample measurement equipment can also capture and measure OM particles >1 mm as part of a VSS sample (USGS 1985). Together, the FPOM and DOM categories comprise about 95 percent of TOM in the Jordan River (Epstein et al. 2016).

Measurements of FPOM (VSS) were collected on a monthly basis at 25 locations, including sites on the Jordan River, outlets of major tributaries, the Surplus Canal, and above and below WRF operations that discharge to the Jordan River (Figure 3-2, Miller 2019b). Annual average VSS concentrations were reported as similar in 2010 and 2011, at approximately 10 mg/L. No statistics were provided, and the large scale of plots makes a close comparison difficult. Monthly VSS concentrations had irregular increases at some locations and general trends at others. Sites at 3300 South and 2100 South both show slight increases March–May and again in the late summer/fall.

Seasonal analysis of total suspended solids (TSS) and VSS data from 2012 included summer (May–October) and winter (November–April). High runoff conditions during early summer were likely responsible for elevated VSS and TSS until the diversion at 2100 South. Although Big and Little Cottonwood Creeks both had lower concentrations of VSS and TSS, concentrations at 3300 South were similar to those measured upstream of tributaries, indicating little dilution.

Measurements of DOM were collected from the LJR upstream of the 1300 South stormwater outfall 2015–2016. Grab samples collected during dry weather and storm events were measured for DOC and 5-day and ultimate soluble cBOD to determine DOM loading and the relative biodegradability of DOM in the LJR compared to stormwater discharge. As indicated by continuous measurements of fDOM, DOM loading by the river made a relatively smaller percent contribution to the total DOM load (i.e., LJR and 1300 South outfall) compared to percent contribution to total flow by the river. Analysis of biodegradability of DOM carried by each flow source indicated significant differences, however. Detailed results are described in section 3.4.

3.3 CHARACTERISTICS OF ORGANIC MATTER

Understanding characteristics of OM that influence the rate of decomposition and corresponding oxygen demand can help prioritize source controls and identify an efficient path to improve water quality in the LJR. OM sources that are highly labile or partially decomposed present more potential for generating oxygen demand compared to sources that are recalcitrant.

Characteristics of DOM can be determined using emission-excitation matrix (EEM) analysis to generate corresponding fluorescence index (FI) values that indicate the source of DOM. Water column measurements of FI in areas without vegetation (e.g., Antarctica) are about 1.8–2.0, much higher compared to trophic rivers (e.g., Suwannee River with intact wetland) that range from 1.1–1.2. Therefore, lower FI values can be associated with DOM from plant matter (including terrestrial and aquatic vegetation), while higher FI values represent microbial biomass or material sourced from microbes (stormwater and/or wastewater).

Summer and Winter VSS in the Jordan River, 2012
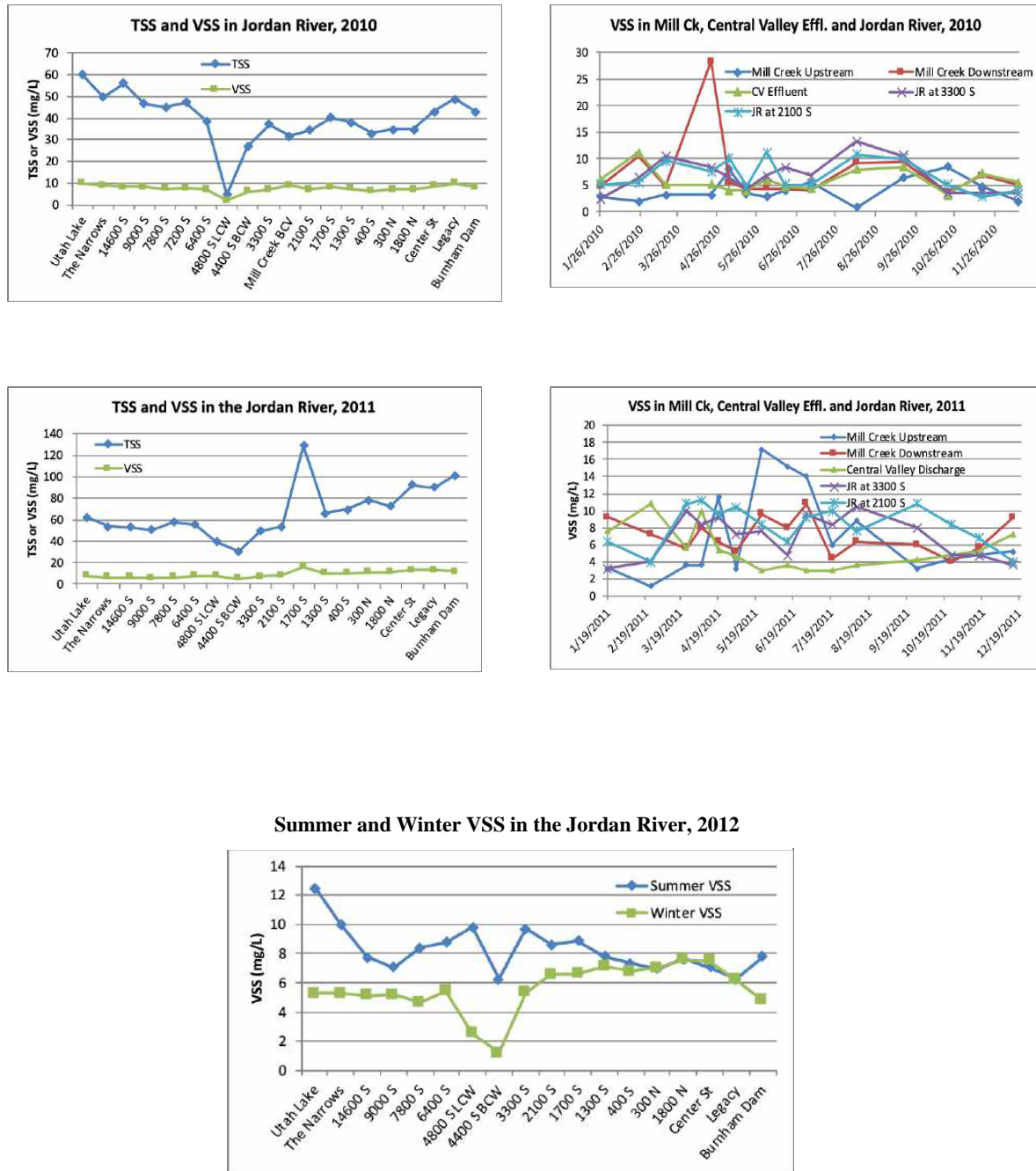
Figure 3-2. Annual average TSS and VSS measured from locations on Utah Lake, Jordan River, and Big and Little Cottonwood Creek in 2010 and 2011 (left side). Monthly average VSS collected above and below CVWRF in 2010 and 2011 (right side). Fifteen samples were collected from each site and included at least one sample event each month. Summer and winter average VSS in 2012 (bottom center) measured from locations on Utah Lake, Jordan River, and Big and Little Cottonwood Creek. Source: Miller (2019b) Figures 62-67.

FI values are very high for the Jordan River (as high as 2.2–2.4 during summer) relative to other aquatic systems, which suggests that microbes and/or microbial residue may comprise a significant portion of DOM in the water column (Follstad Shah et al. 2017). However, EEMs have not been highly used in urban rivers to analyze DOM. Urban rivers could potentially include constituents that modify FI values compared to watersheds without urban influences. It has been recently discovered that different laboratory equipment used to calculate EEMs can produce different values. These variations are currently being investigated to quantify the differences between machine outputs using a consistent set of samples for analysis (Follstad Shah 2020). Although comparison of FI values to other systems should be made with caution, relative changes in FI values along the Jordan River flowpath can be compared between study sites and provide meaningful information.

Elevated FI values downstream of WRFs relative to upstream sites in all seasons indicate that WRF inputs (as high as 2.8) influence OM composition in the Jordan River (Figure 3-3). Elevated FI values in the fall were compared to other seasons, which suggested that effluent is the dominant source of water in the river during this time of year. The previous synthesis (Cirrus 2017a) included isotope analysis results that suggested the annual DOM load was primarily (90 percent) from terrestrial sources. These results did not account for seasonal variation.

Measurements of ecoenzyme activity in the Jordan River can also be used to characterize the nature of OM (Follstad Shah et al. 2017, Follstad Shah 2019b). Comparing ecoenzyme activity rates between those associated with labile C source (BG) metabolism to those associated with recalcitrant C source (POX) metabolism provides useful information on the nature of Jordan River OM. Results show BG:POX ratios indicative of much greater microbial activity related to labile OM sources (i.e., glucose) than to recalcitrant OM sources (i.e., lignin; Figure 3-4). These measurements were collected from unfiltered samples and thus characterize both dissolved and particulate forms of OM. These results indicate that the microbial community is utilizing more labile forms of polysaccharides relative to more recalcitrant forms of carbon, such as lignin and tannins.
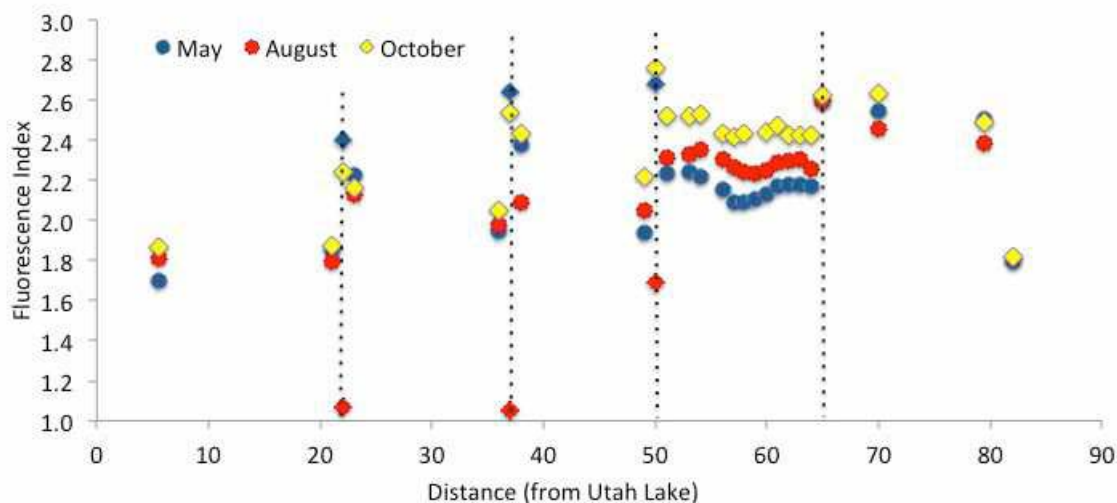
Figure 3-3. Fluorescence Index (FI) values of dissolved organic carbon (DOC) measured from the Jordan River and water reclamation facilities (diamond symbols that appear on dashed vertical lines) in spring (blue), summer (red), and fall (yellow). FI values (y-axis) are derived from excitation-emission matrix (EEM) analyses. Values on the x-axis indicate downstream distance (km) from Utah Lake. Source: Follstad Shah et al. (2017) Figure 10.
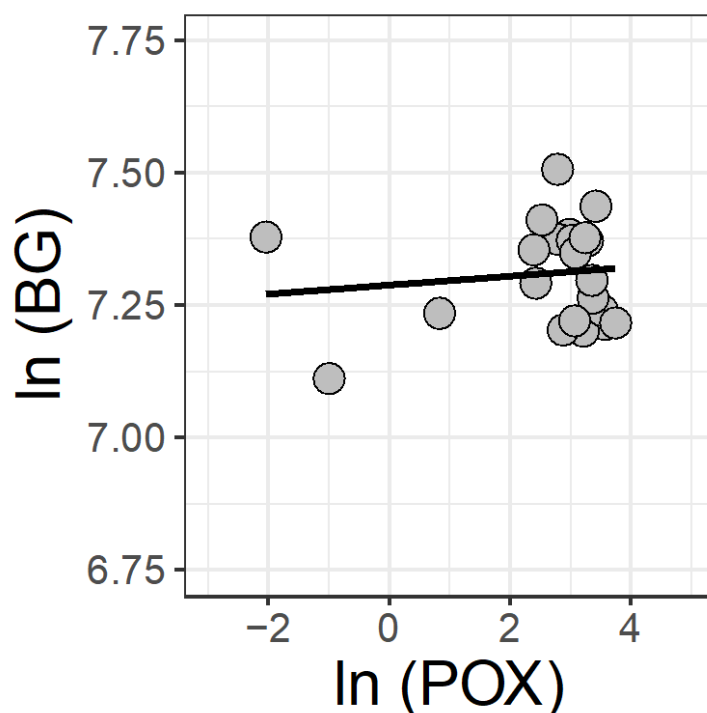
Figure 3-4. Microbial activity (nmol/h-L) for BG and POX ecoenzymes used to acquire C from the Jordan River water column. Results show much higher activity rates from BG compared to POX (~2 orders of magnitude more), reflecting acquisition of C from labile sources (i.e., glucose) relative to recalcitrant sources (i.e., lignin). Source: Follstad Shah et al. (2019b).

Sites directly receiving WRF effluent had OM with significant WRF characteristics including microbial derived, protein-like DOM (Kelso 2018).

3.4 ORGANIC MATTER SOURCES

The previous synthesis document identified loading estimates for Jordan River OM that included DOM, FPOM, and CPOM (Epstein et al. 2016). Stable isotopes of C (δ^{13}C), N (δ^{15}N), and H (δ^2H) concentrations have been measured at nine sites on the Jordan River and Mill Creek and used as tracers to identify OM source contributions to each size category (Kelso 2018). Five source types were evaluated for each size category of OM: terrestrial sources, aquatic sources, benthic organic matter (BOM), WRF effluent, and Utah Lake.

Terrestrial sources included leaf-litter (senesced), tree leaves (not senesced), and *Phragmites*. Aquatic sources included macrophytes, biofilm, and algae. Macrophyte samples were cut from large submerged aquatic vegetation anchored in the benthic zone. Biofilm was scraped from benthic rocks, and algae were collected from green mats floating on the water surface. WRF effluent and lake sources were collected for all three size classes. Samples were collected from sources during fall, winter, spring, and summer months.

Two models were used to analyze stable isotope results. The Stable Isotope Mixing Model in R (SIMMR) is a Bayesian mixing model that estimates the proportional contribution of sources to a mixture regardless of the number of isotope tracers. This model was designed to estimate the proportional contribution of sources as they appear in combination or mixture. A graphical, gradient-based mixing model was also used to partition OM sources as either terrestrial or aquatic.

Analysis of CPOM data from the stable isotope and SIMMR modeling approach resulted in three likely sources, with leaf-litter being the most dominant source except during summer months, when macrophyte contributions were roughly equal to leaf-litter (Figure 3-5). Biofilm was the least likely source of CPOM with relatively more contribution occurring in the spring (3–34 percent) and summer (5–26 percent) seasons compared to the fall and winter seasons.

FPOM results showed variation across all months sampled (Figure 3-6). Terrestrial sources tended to be higher in the fall, while BOM and Lake FPOM were higher contributors to the total FPOM pool in the summer. Concentrations of FPOM from WRFs increased beginning in September and continuing through November.

DOM has three likely sources including Lake DOM, WRF DOM, and litter and tree leaves (Figure 3-7). The largest contributor to the DOM pool was Lake DOM, with an average contribution of 57 percent over the course of the year and a median range of 48–70 percent. WRF DOM had an estimated average annual contribution of 27 percent with a median range of 20–33 percent in all months. Terrestrial contributions averaged 16 percent annually, with a maximum mean value of 29 percent occurring in September.

Both models indicated that Lake DOM was the primary source of DOM in the Jordan River and that DOM had major contributions from WRFs throughout the year.
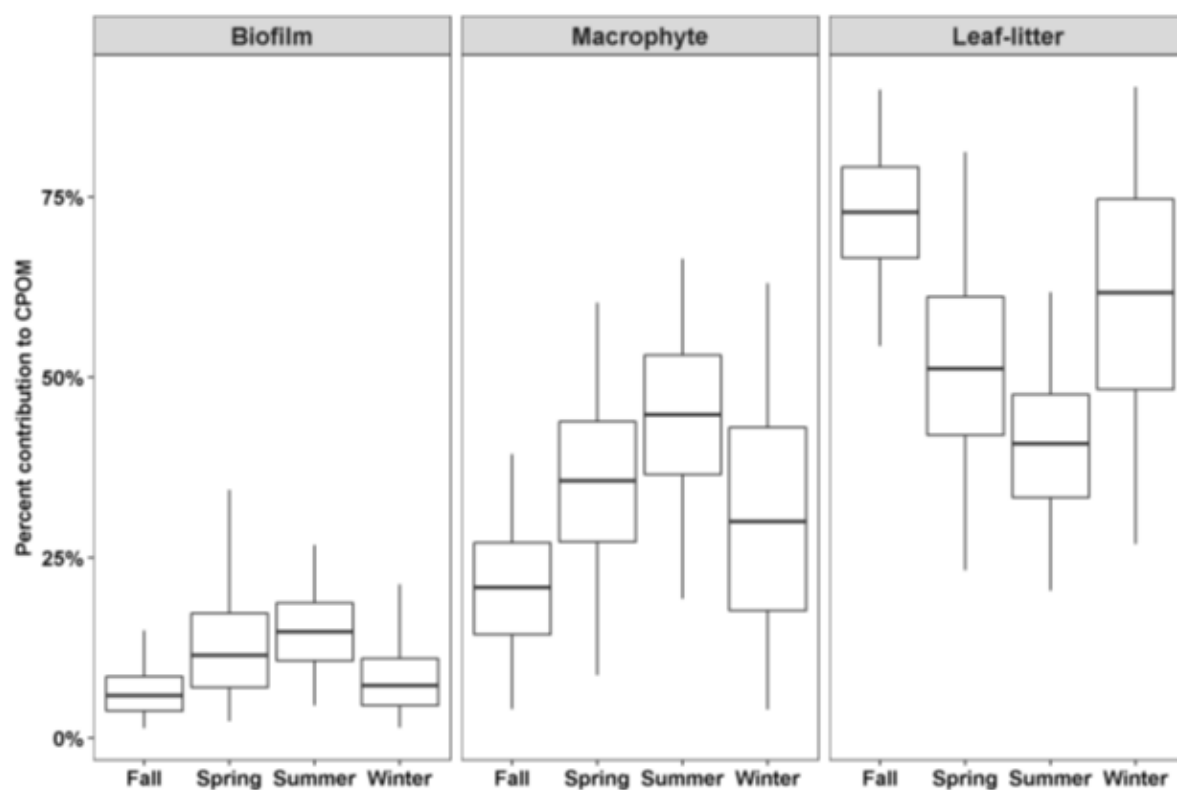
Figure 3-5. Percent CPOM contribution estimates of three sources as a function of season. Contributions were estimated using the SIMMR mixing model with three isotope tracers δ^{13}C, δ^{15}N, and δ^{2}H. Boxes represent the median and 75 percent high-density interval; whiskers represent the 95 percent high-density interval. Source: Kelso (2018) Figure 3.
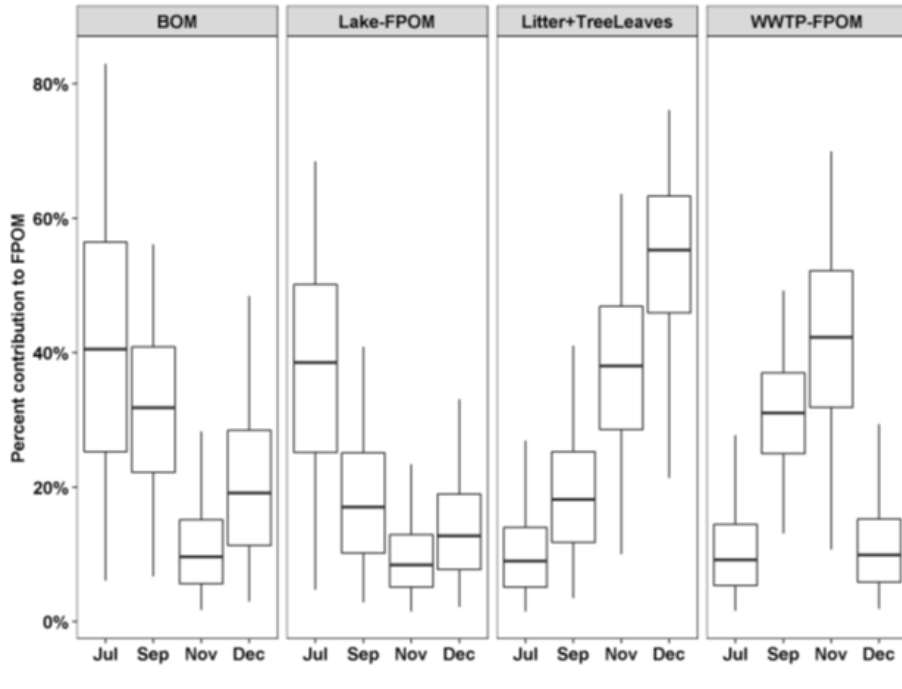
Figure 3-6. Percent FPOM contribution estimates from four sources as a function of season. Contributions were estimated using the SIMMR mixing model with three isotope tracers δ^{13}C, δ^{15}N, and δ^{2}H. Boxes represent the median and 75 percent high-density interval; whiskers represent the 95 percent high-density interval. Source: Kelso (2018) Figure 4.
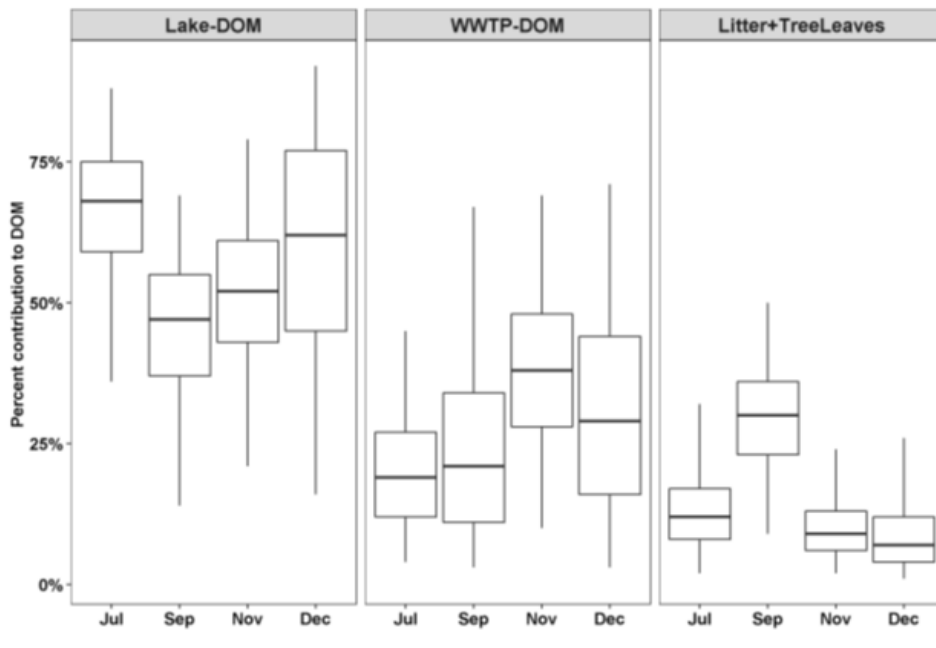
Figure 3-7. Percent DOM contribution estimates from three sources as a function of season. Contributions were estimated using the SIMMR mixing model with two isotope tracers δ^{13}C and δ^{2}H. Boxes represent the median and 75 percent high-density interval; whiskers represent the 95 percent high-density interval. Source: Kelso (2018) Figure 5.

Isotope tracing (δ^{15}N) was used to determine if FPOM in wastewater effluent is a source of N for instream biota. Results showed that the δ^{15}N of FPOM in wastewater treatment plant effluent was quite variable with respect to the Jordan River (Follstad Shah et al. 2017). Effluent from the JBWRF had consistently lower values than the Jordan River, while effluent from CVWRF had consistently higher values than the river. Effluent from SVWRF had δ^{15}N in FPOM lower than the Jordan River in spring and fall but higher in the summer. Although δ^{15}N FPOM signatures could not be correlated with effluent discharge, values downstream of CVWRF were consistently high in the river indicating its consistent influence on FPOM signatures in the Jordan River. This response could be due to differences in technology used at each WRF discharging to the river.

No distinction was noted in recent δ^2H values of biofilms and riparian vegetation samples collected from the Jordan River (Follstad Shah et al. 2017). Recent FPOM δ^2H values were also found to be like other values measured in 2013 (Kelso 2017), as noted in the previous synthesis report (Figure 3-8). Differences in flow may have altered the relative contributions of terrestrial and aquatic sources during these 2 years, but it is currently not possible to distinguish between different sources without distinct signatures from biofilm and riparian vegetation.

Chemical and biological characteristics of OM carried by stormwater were analyzed by Richards (2015) in the previous synthesis review. Recent stormwater research builds on that work to document long-term OM contributions of stormwater C loading to oxygen impairment in the LJR (Dupont et al. 2018).

The 1300 South storm drain to the Jordan River receives flow from Red Butte Creek, Emigration Creek, and many stormwater catchments in Salt Lake City east of the Jordan River. This outfall and the Jordan River at 1700 South are continuously monitored for flow and a suite of water quality parameters, one of which is fDOM. ISCO automated grab samples were collected from the stormwater channel and the Jordan River during storm events and dry periods to develop correlations between DOC and BOD (5-day and ultimate soluble cBOD) along with continuously monitored fDOM surrogate measurements.
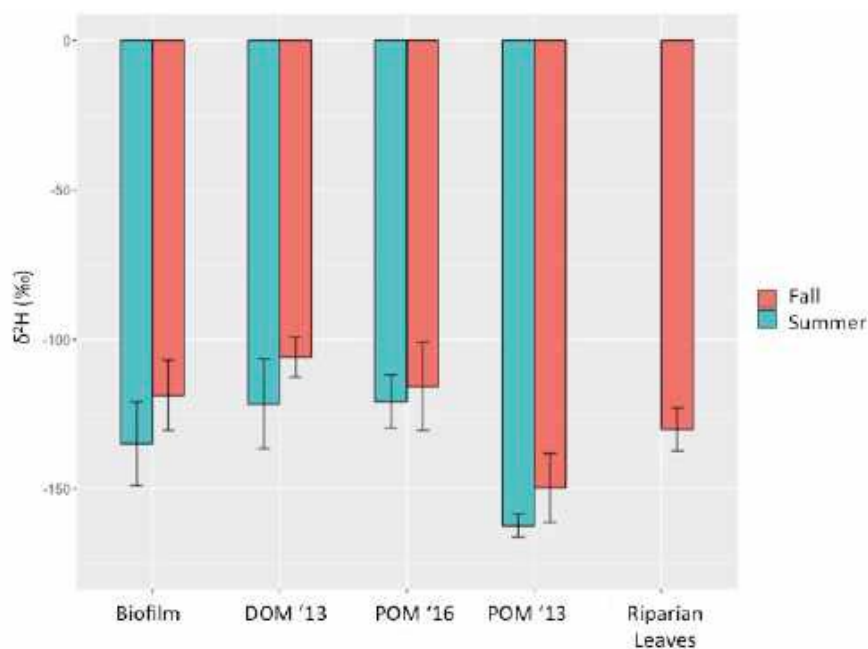
Figure 3-8. Similarity in δ^2H values in summer (blue) and fall (red) for biofilms, DOM (measured in 2013), FPOM (measured in 2013 and 2016) and riparian leaves. Measurements from 2013 are from Kelso and Baker (2017). Source: Follstad Shah et al. (2017) Figure 9.

Continuously monitored flow data from the Jordan River and 1300 South outfall measured June 1, 2015–June 30, 2016, were screened and paired to produce 35,003 paired data points. Results indicated the outfall represents about 6 percent of Jordan River flow on average, with summer discharges that increase to 20 percent, and intermittent extreme storm events that exceed 150 percent of total Jordan River flows.

Continuously monitored water quality data were also screened and paired during the same period to produce a total of 19,002 data points. If fDOM measured in the LJR has the same biodegradability and oxygen consumption characteristics as fDOM from the outfall, the average stormwater fDOM loading contribution is on average, 3.2 percent of the fDOM load in the Jordan River at 1700 South. However, results showed that fDOM stormwater loads from this 1300 South outfall frequently peaked above 30 percent and exceeded 180 percent of the upstream Jordan River fDOM load during a storm in September 2015.

Grab samples were tested to determine the relative biodegradability of DOC from each source and to develop BOD/fDOM relationships for the 1300 South outfall and the upstream Jordan River samples. The decay rates of BOD were found to be similar for the 1300 South outfall and the Jordan River, but the relative degradability of fDOM, as indicted by the BOD/fDOM ratio, was significantly different for the two flow streams. The BODu/fDOM ratio of organics released to the Jordan River from the 1300 South outfall was found to be 0.20±0.07, compared to 0.03±0.01 for the Jordan River upstream from the outfall. The organic material being released to the Jordan River from this outfall was significantly less stable, representing significantly more oxygen demand, than the fDOM passing this discharge point in the Jordan River.

Paired water quality data sets were again used to calculate the percent BODu loading of the 1300 South stormwater outfall, normalized to the upstream Jordan River loading at 1700 South, accounting for the difference in relative degradability of the fDOM in each flow stream. Figure 3-9 indicates that BODu loading from stormwater exceeds 1,200 percent of the Jordan River background load when the relative degradability of DOC (as indicated by the fDOM/BODu relationship) is accounted for. Despite the fact that 1300 South contributes 3.2 percent (on average) to the total fDOM load, the high biodegradability of fDOM generated by stormwater causes an average ratio of >26 percent with routine excursions >100 percent even during small storms (Figure 3-9). These results clearly indicate that stormwater discharge to the LJR is a significant and on-going intermittent source of oxygen demand that contributes to impairment.

Corbicula can biologically function as both a sink and a source of OM in the Jordan River. These macroinvertebrates remove sediment OM by pedal feeding with cilia on their foot to gather buried OM (Richards 2018). *Corbicula* can also contribute OM to channel sediments by compositing small seston particles into larger aggregates of feces and pseuodfeces in a process known as biodeposition. Consumption of larger POM particles favors production of pseudofeces when particles are enveloped in mucous and ejected as partially or undigested POM. Settling rates of this material can be up to 40 times faster than non-aggregated particles, although pseudofeces may disaggregate when voided. However, mucous surrounding pseudofeces is also known to function as a binding agent for sediment that limits resuspension processes. *Corbicula* biodeposits have been noted to contain 2–3 times more C, N, and P per unit weight as particles that settle out naturally from the water column.

3.5 FUTURE PREDICTED CHANGES IN ORGANIC MATTER

The Phase 2 TMDL will need to include an allowance for increases in pollutant loading due to changes in land use, flow, and other factors that could reasonably change and influence water quality in the future. Potential changes in the magnitude and timing of streamflow and sediment yield were estimated for the 2040s (Year 2035–2044) and 2090s (Year 2085–2094) using the HSPF flow and water quality model and different scenarios for climate and land use/land cover (Khatri et al. 2019). Three climate scenarios were downscaled from global climate models. Future change in land use and land cover were determined using a regional planning model and a deterministic model.
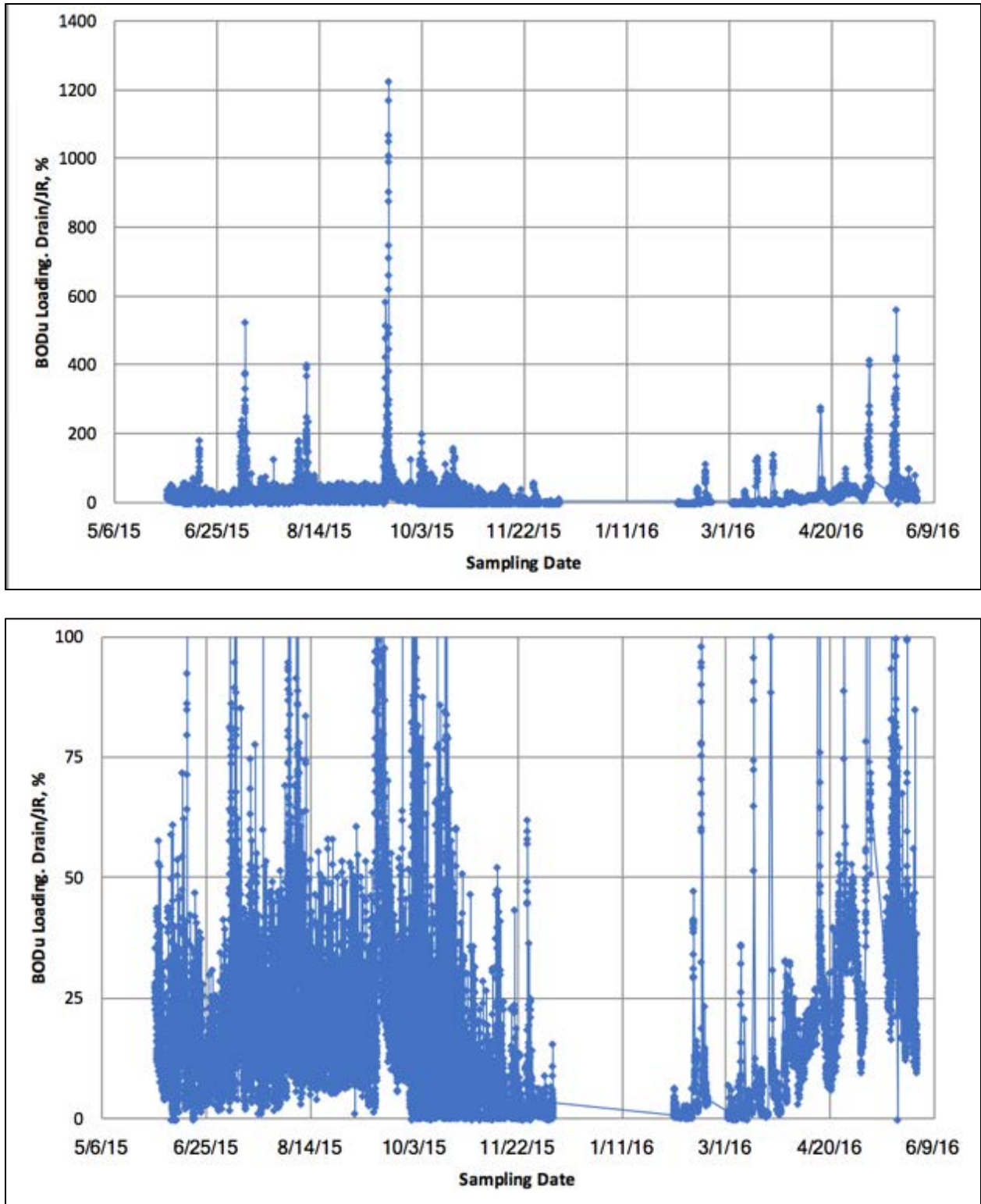
 Figure 3-9. Ratio of stormwater channel BODu loading to LJR BODu loading as a percent during the study period June 1, 2015–June 30, 2016. Lower plot shows an expanded y-axis from 0–100 percent for the same data set. Source: Dupont et al. (2018) Figures 9 and 10.

Results were reported for Big Cottonwood Creek (BCC) and the canyon mouth (climate change only, no change in land use/land cover) and the Jordan River above Surplus Canal diversion (climate change and land use/land cover change). Model results focused on a change in date of the 50th percentile cumulative annual streamflow to assess impacts on the timing of snowmelt-dominated runoff and sediment loading. Land use scenarios included: 1) continuing existing land use/land cover patterns, 2) business-as-usual growth, and 3) centers-oriented growth. The business-as-usual growth scenario was developed by the Wasatch Front Regional Council and incorporates logistic regression models for five land-uses including retail, other employment, industrial, single-family residential, and multi-family residential. The centers-oriented growth scenario was developed for the metro region's transportation plan. This scenario applies one-third of projected growth into development centers that occupy 3 percent of the growth aerial footprint.

Modeled results indicate mean daily streamflow in the Jordan River watershed is predicted to increase (as compared to historical records of the 2000s) by an amount ranging from 11.2 percent (BCC) to 14.5 percent (Jordan River at Surplus Canal) in the 2040s and from 6.8 percent (BCC) to 15.3 percent (Jordan River at Surplus Canal) in the 2090s (Table 3-2). The predicted increases in sediment loads in the 2040s and 2090s are projected to be 6.7 percent and 39.7 percent, respectively, in the canyons and about 7.4 percent to 14.2 percent in the Jordan River, respectively. The historical 50th percentile timing of streamflow and sediment load is projected to be shifted forward by 3 to 4 weeks (into April) by mid-century and 4 to 8 weeks (into March) by late-century.

The study projected that mean daily sediment loads for BCC would increase 1.4 tons/day by the 2040s and 1.7 tons/day by the 2090s. The significant increase in sediment loading predicted for this tributary stream was due in part to the predicted increased fraction of rain versus snow and more frequent and higher-intensity rainfall events intercepted by a steeply sloped watershed. The relative increase in future daily sediment loads in the Jordan River was less pronounced than in BCC.

The shift toward earlier timing of the 50th percentile of the mean annual flow is expected to reach about 4 weeks in the 2040s and 8 weeks in the 2090s, with an uncertainty of 4 and 2 weeks, respectively, depending on a given climate scenario. Study results suggest that the range of possible future stream flows and sediment loads in the Jordan River watershed is primarily driven by future climate change, whereas the changes in land use/land cover play a secondary role.

In general, the impacts of warming climate and increased precipitation may induce seasonal shifts that could alter patterns of streamflow and sediment loading. The early shift in the timing of the 50th percentile in the streamflow and sediment loading is driven by changes in precipitation, snowpack, and snowmelt, with a nonlinear relationship between streamflow and the sediment load.

3.6 INFLUENCE OF CHANNEL FEATURES ON OM LOADING

The LJR is a low-gradient channel prone to deposition along much of its length. Flow management creates additional impacts on transport of material in the water column and as bedload in these river segments. Restoration projects in segments of the Jordan River upstream of 2100 South have created bank and channel features that can transport sediments and build healthy riparian systems that function to help minimize flood damage to the banks of the river.

More than 3,600 lineal feet of Jordan River from 5100 South to 4800 South were reconstructed from October 2015–November 2018 using natural channel designs including toe wood structures (Salt Lake County 2018). Cross-section measurements collected before and after restoration show decreases in bank-full width (from 85 to 73 ft), bank-full depth (from 2.95 to 2.61 ft) and bank-full area (from 251 to 190 ft). Before and after pebble counts identified silt, clay, and finer sands deposited at water's edge where a natural floodplain is beginning to develop. Pebble counts also showed that finer sediments were no longer a predominate feature in bed sediments or along inner slopes leading to the thalweg. Visual observations verified that silts and finer sands are being deposited on the floodplain or being moved downstream by

increased velocities in the deeper thalweg to a detention basin which helps improve the sediment transport mechanism in the restored segment.

Improvements to riparian and upland areas, including active floodplains, will increase resilience to future flood events. Restoration has also increased the river's ability to maintain healthier DO concentrations and water temperatures by providing shade along the riverbanks and filtering sediment and nutrient loads before they can enter the river. Estimated load reductions include 111.6 tons/year of sediment, 188.2 lb/year of N, 70.4 lb/year of P, and 358.4 lb/year of BOD. As the restoration continues, the total reduction of sediment, N, and P will be significantly greater (Salt Lake County 2018).

Table 3-2. Projected future change in streamflow and sediment concentration under three climate change scenarios including minimum downscaled climate change projections, moderate greenhouse gas emissions (RCP6), and maximum downscaled climate change projections.

Climate Change Scenarios/parameters	Historical 2000s	Minimum		RCP6		Maximum	
		2040s	2090s	2040s	2090s	2040s	2090s
Streamflow and sediment concentration for Big Cottonwood Creek at canyon mouth.							
A-Streamflow (m 3/8)							
Mean	2.1	1.8	2.0	2.4	2.3	2.1	2.6
Minimum	0.7	0.6	0.7	0.7	0.6	0.6	0.7
Maximum	9.5	8.2	8.7	7.5	9.4	9.0	9.3
Standard deviation	2.1	1.9	2.0	1.9	1.6	2.0	2.2
B-Sediment concentration (mg/L)							
Mean	3.4	3.3	3.4	4.5	6.2	5.0	6.3
Minimum	1.5	1.5	1.5	1.5	1.4	1.5	1.6
Maximum	24.9	47.0	33.4	41.5	106.7	62.6	60.4
Standard deviation	3.5	3.9	3.8	4.7	9.5	9.5	8.2
Streamflow and sediment concentration for Jordan River above Surplus Canal.							
A-Streamflow (m 3/8)							
Mean	13.8	14.1	14.7	15.8	15.9	15.5	15.9
Minimum	6.8	7.8	8.1	8.0	8.1	7.9	8.1
Maximum	32.8	27.8	30.3	30.6	32.7	36.3	32.8
Standard deviation	6.0	5.0	5.2	5.9	5.3	6.0	5.3
B-Sediment concentration (mg/L)							
Mean	28.7	28.8	28.2	27.6	29.7	29.1	29.3
Minimum	1.4	0.6	0.6	0.6	0.5	0.7	0.8
Maximum	72.8	62.7	62.2	91.9	160.6	95.4	99.9
Standard deviation	18.2	16.4	16.0	15.7	18.8	16.8	16.9
Source: Khatri et al. (2019) Table 2 (Big Cottonwood Creek) and Table 4 (Jordan River).							

It is highly likely that restoration projects like this can help restore the upper and lower Jordan River to a more natural state while still maintaining the flood-control aspects of the river and supporting its beneficial uses. A critical aspect of restoration in the LJR will be increasing baseflow conditions and promoting a more natural hydrograph where possible (Thompson 2020).

Some features of the 2100 South diversion have been discussed previously (Section 2.1.4) that potentially influence OM loading to the LJR. This structure provides for a bottom release of water into the LJR and could preferentially capture bedload OM instead of allowing this load to pass into the Surplus Canal. Redesigning this structure to provide a top-release of water could eliminate the potential for capturing bedload OM as well as provide additional reaeration to the LJR. This change would also improve the forebay above the existing diversion and create a more effective settling basin (Miller 2019c).

A second suggested modification is to elevate the upper segment of the LJR to increase the LJR channel gradient from 2100 South downstream to the PacifiCorp diversion (Miller 2019c). Several sedimentation basins could be included in the restored segment to remove transported material that would normally contribute to SOD in downstream segments. The restored channel segment could be covered with cobble and gravel to improve habitat, increase surface roughness and natural reaeration and provide improved processing of organic material that contributes to SOD.

This page is intentionally left blank.

4.0   JORDAN RIVER WASP IMPLEMENTATION ASSESSMENT

This section includes a review of the *Jordan River Modeling through the Water Quality Assessment Simulation Program (WASP): Model Report* (Su 2019). Development and application of the Jordan River WASP model is a critical piece of completing the Phase 2 TMDL. Water-quality modeling is needed to characterize and define important processes that influence acute and chronic DO in the LJR. Model results can also be used to quantify relationships between pollutant sources and water quality in support of load allocations.

The Jordan River WASP model was developed as part of a larger project to assess the Jordan River watershed under existing and future climate change conditions and land use projections (Barber et al. 2016). Specific to the Jordan River, the WASP modeling effort has five objectives that include:

1. Extend previous studies conducted over the Jordan River from the steady-state Qual2K models to the unsteady-state/dynamic continuous simulations.
2. Assess the water quality performance of the Jordan River over time, primarily over the summer months under elevated water temperatures and low flows.
3. Develop and suggest potential linkages among the water-quality performance of the Jordan River against the effluent quantity and quality from Utah Lake.
4. Develop and suggest potential linkages among the water-quality performance of the Jordan River subject to historical and potentially futuristic land use, such as urbanization.
5. Develop and suggest potential linkages among the water-quality performance of the Jordan River subject to historical and potentially futuristic climate characteristics (e.g., climate change projections through the representative concentration pathways (RCPs), etc.)

The results discussed in this section indicate progress toward these objectives.

4.1 MODEL DEVELOPMENT

The structure of the Jordan River WASP model is similar to the original QUAL2kW model developed for the Jordan River TMDL process (Stantec 2010). The WASP model includes a total of 166 surface segments covering 51.8 miles that begin at Utah Lake and end at Burton Dam, about 4.6 miles below the South Davis South Water Reclamation Facility (SDSWRF). Inflows and outflows include effluent from four WRFs, nine tributary streams, and six major stormwater outfalls. Time-series data are required as inputs; parameters that were not available were interpolated between available data points. The model was implemented over a period of 5 water years (October 1, 2006–September 30, 2011).

A sensitivity analysis was conducted using two methods to assess the performance of the WASP model. One method tested model inputs from measured data or previous models (e.g., QUAL2kW) by changing (increase/decrease) the parameter value by factors of 2, 10, and 100. The second method tested model parameters that were not retrieved from previous models or published literature. These parameters were tested by changing the parameter to a particular value rather than a percentage.

Analysis results showed that several parameters were either generally sensitive or significantly sensitive, which indicates their ability to alter the performance of constituents that the parameter corresponds to (e.g., phytoplankton growth upon phytoplankton chlorophyll-a). These parameters were primarily related to heat exchange and temperature of bottom sediments, algae production of O_2 and C, and classification of particulate OM into labile, refractory, or inert categories.

The WASP model was calibrated using a graphical and a statistical approach. Modeled estimates of water quality and flow were compared to the measured data for river segments where monitoring data was

collected. Six different statistical parameters were calculated based on differences between measured and modeled data. Both approaches were used to evaluate model performance after adjusting select input parameters (based on sensitivity analysis). Model calibration primarily relied on minimizing mean absolute error and the root-mean-square error for all WASP segments compared with measured data. No auto-calibration tools or approaches were used for model calibration.

4.2 MODEL CALIBRATION RESULTS

Model calibration results include the following important points (Su 2019):

- Modeled DO results generally agree in the UJR above Turner Dam. From Turner Dam down to 5400 South, DO appears to be underpredicted, particularly prior to Water Year 2009 (October 1, 2009). The model appears to slightly overpredict the DO concentration in the LJR as compared to measured data.

- Overprediction of DO in the LJR may be due to simulated SOD values produced by the sediment diagenesis routine. Simulated SOD values appear to be significantly lower than those observed by Hogsett (2015). Additional investigations are needed to adjust model input parameters that simulate the sediment diagenesis routines to increase modeled SOD values, particularly downstream of the Surplus Canal.

- Modeled DO concentrations are heavily influenced by the model input parameter O_2:C production by macro algae. The observed dependence of DO on macro/benthic algae appears to be the reason for overpredicting DO, which suggests there are underlaying processes that appear to not be captured by the WASP model.

- General agreement is evident between model estimates of CBOD and measured BODu records from DWQ. These results may require more analysis due to the quality of measured data due to the significant number of non-detect values in the data set.

- Except for a few outliers in some segments, model estimates of phytoplankton/chlorophyll-a agree with measured values.

- Regarding nutrients, model estimates of total phosphate generally agree with measured data. Both ammonia (as NH_3-N) and nitrite-nitrate (NO_2-NO_3-N) are underpredicted in the water column, as is DON below 5400 South.

- Modeled sediment- ammonia benthic flux produced by the WASP sediment diagenesis routine is substantially lower than measurements by Hogsett (2015), suggesting that some processes influencing flux may not be captured by the WASP model.

The remainder of this section includes guidance for improving the Jordan River WASP model and using the model to determine load allocations for the Phase 2 Jordan River TMDL.

4.3 GUIDANCE FOR FUTURE JORDAN RIVER WASP APPLICATIONS

In reviewing the documentation provided for the Jordan River WASP application (Su 2019), several items have been identified that would strengthen this modeling application and make it more relevant for future management decisions in the Jordan River. In general, application of any water quality model to a river system should first ensure that flow volumes and flow rates are represented well. Once this is complete, the temporal and spatial variability of water temperature should also be checked, as temperature controls all reaction rates. High-frequency temperature data are relatively easy to collect and model testing against these data can provide additional insight regarding 1) the importance and representation of point and nonpoint inflows throughout the model domain and 2) representativeness of channel properties used within the model because air water interface heat fluxes are scaled by surface area. Next, the mass loading rates (or volumetric flow rate multiplied by the concentration) of various constituents for different inflows (point and distributed) should be revisited. In urban watersheds with a variety of large point and nonpoint inflows,

these loads are large enough that they can significantly influence, or even reset, the quality of the river over time and space.

If the flow rates throughout the study area are representative based on the first step outlined here, then this activity should focus on determining the concentration of the relevant constituents in these inflows over time. Finally, the last step is calibrating sensitive parameters that control the fate of constituents over time and space. The following sections provide some guidance on more specific steps that should be considered for future use of the Jordan River WASP application.

4.3.1. Flow balance and Routing

Su (2019) based the Jordan River WASP model segmentation and hydraulic parameterization on prior Jordan River QUAL2Kw modeling that used information from prior flood prediction tools. Given this history and the need to focus on low flow periods (see sections below), there is a need to determine the accuracy of the Jordan River WASP application when it comes to representing time variable flow volumes (particularly during low flow periods). This is critical because there are inflows that are not currently being accounted for in this application (see Table 6 in Su 2019) and it is unclear which inflows are critical for capturing water quality responses at the scales of interest. Additionally, there is a need to test the assumptions made regarding groundwater inflow over long timescales.

There is also a need to determine if the model is correctly routing flows through the model domain. The Jordan River WASP model currently uses the kinematic wave flow routing method, which is likely an appropriate choice, but the hydraulic data used in model set up from prior studies may need to be refined. More specifically, it is important to know if WASP model results accurately represent the residence times and velocities in different river segments. This effort could focus on representative rain events during seasons that are critical for transporting sediment and both internal (upstream algal transport) or external OM loads to segments of the LJR where DO is a concern.

4.3.2. Temperature

The two most sensitive parameters identified in Su (2019) were the sediment temperature (T_s) and the coefficient of sediment/water heat exchange (K_{sw}). In the WASP8 Temperature Model Theory and User's Guide (Wool et al. 2008), they provide the following equation for the rate of sediment/water heat exchange.

$$H_{sw} = -K_{sw} (T_w\text{-}T_s) \qquad\qquad Eqn\ 1$$

Where:

H_{sw} = rate of sediment/water heat exchange, W m^{-2}
K_{sw} = coefficient of sediment/water heat exchange, W m^{-2} °C^{-1}
T_w = water temperature, °C
T_s = sediment temperature, °C

The heat flux associated with sediment conduction, using similar variables as those in WASP, could be represented by

$$H_{sw} = -K (T_w\text{-}T_s)/dz \qquad\qquad Eqn\ 2$$

Where:

K = thermal conductivity (Wm^{-1}°C^{-1})
dz = distance or depth over which the temperature gradient (T_w-T_s) occurs (m)

When examining Eqn 1 and 2 together, the WASP K_{sw} value appears to represent the thermal conductivity (K) divided by some distance or depth (dz).

Based on the values of thermal conductivity (K in Eqn. 2) provided within the QUAL2Kw manual (Table 1, Pelletier and Chapra, 2008), values for typical bed sediments range from 0.36-2.5 $Wm^{-1}°C^{-1}$). A statement in Wool et al. (2008) reported, "Cole and Buchak (1994) indicated that values of 7 x 10^{-8} W m^{-2} $°C^{-1}$ for K_{sw} have been used in previous applications and that average yearly air temperature is a good estimate of T_s." Because this K_{sw} value from Cole and Buchak (1994), a prior CE-QUAL-W2 manual, was very low, a more recent CE-QUAL-W2 manual (Cole and Wells 2003) was found. This updated version changed this statement to "Previous applications used a value of 0.3 $W m^{-2} °C^{-1}$ for K_{sw} that is approximately two orders of magnitude smaller than the surface heat exchange coefficient. Average yearly air temperature is a good estimate of T_s." The latter value of K_{sw} seems reasonable for lake sediments (0.6 $Wm^{-1}°C^{-1}$) and a sediment temperature (T_s) representative of the annual average air temperature at $dz = 2$ m depth.

In the Jordan River WASP application, $K_{sw} = 10$ $Wm^{-2}°C$ and $T_s = 14$ °C. While 14 °C is a reasonable annual average air temperature for this area, the K_{sw} is not a reasonable value. This would suggest that the sediments are a large heat sink in the Jordan River WASP application and will likely result in lower temperatures than observed. However, if there are cooler inflows that are not accounted for, this heat loss to the sediments may just be compensating for these.

The ranges tested in the sensitivity analyses for K_{sw} (0.6 to 6000 Wm^{-2}) and T_s (0-30 °C) included reasonable values. However, the ranges tested were not reasonable. A sensitivity analysis is meant to help understand how the model responds to extreme values, however, they need to be the extreme values of feasible ranges. This is further addressed in the Sensitivity Analysis/Calibration section below.

4.3.3 Shading

Based on results from Neilson (2011), relatively large changes in shading did not result in large changes in temperature along the Jordan River. Therefore, it is likely that the relatively slow changes in in riparian cover along the Jordan River are unimportant in model development relative to the need for capturing the variability in various inflows along the study reach.

Additional testing of the model representation of the upper reach temperatures could use the detailed temperature and flow data collected as part of the Jordan River temperature modeling efforts (Neilson 2011). This would allow a reasonable test of the influence of some current assumptions within the Jordan River WASP application that were necessary due to data limitations (see Table 6 and 7 in Su 2019). Additionally, some of the more recent, longer term sonde data throughout the entire study reach should be used to test the model set up, assumptions, and calibration. (Table 4-1)

4.3.4 Load Estimation

Given the limited headwater, point inflow, and distributed inflow quality and quantity data, there appears to be a need to determine better methods for filling gaps in the observed data. Based on the report, it seems that all values for inflows are linearly interpolated for both flow and water quality and the length of time between samples can be quite long. The frequency of observations versus the actual variability in each inflow should be considered when determining methods for estimating external loads. Initial testing of flow along the Jordan River can help determine how best to handle the quantity issues and assumptions. However, when considering quality, all data available from each point and non-point source (including periods outside of the model simulation) should be considered when establishing the method for handling the temporal variability for each source. Jordan River high frequency data can also assist in testing assumptions regarding the time variability of loads via time variable mass balance techniques with conservative constituents (e.g., specific conductivity).

Type of material	thermal conductivity		thermal diffusivity		ρ	C_p	ρCp	reference
	w/m/°C	cal/s/cm/°C	m^2/s	cm^2/s	g/cm3	cal/(g °C)	cal/(cm^3 °C)	
Sediment samples								
Mud Flat	1.82	0.0044	4.80E-07	0.0048			0.906	(1)
Sand	2.50	0.0060	7.90E-07	0.0079			0.757	"
Mud Sand	1.80	0.0043	5.10E-07	0.0051			0.844	"
Mud	1.70	0.0041	4.50E-07	0.0045			0.903	"
Wet Sand	1.67	0.0040	7.00E-07	0.0070			0.570	(2)
Sand 23% saturation with water	1.82	0.0044	1.26E-06	0.0126			0.345	(3)
Wet Peat	0.36	0.0009	1.20E-07	0.0012			0.717	(2)
Rock	1.76	0.0042	1.18E-06	0.0118			0.357	(4)
Loam 75% saturation with water	1.78	0.0043	6.00E-07	0.0060			0.709	(3)
Lake, gelatinous sediments	0.46	0.0011	2.00E-07	0.0020			0.550	(5)
Concrete canal	1.55	0.0037	8.00E-07	0.0080	2.200	0.210	0.460	"
Average of sediment samples:	*1.57*	*0.0037*	*6.45E-07*	*0.0064*			*0.647*	
Miscellaneous measurements:								
Lake, shoreline	0.59	0.0014						(5)
Lake soft sediments			3.25E-07	0.0033				"
Lake, with sand			4.00E-07	0.0040				"
River, sand bed			7.70E-07	0.0077				"
Component materials:								
Water	0.59	0.0014	1.40E-07	0.0014	1.000	0.999	1.000	(6)
Clay	1.30	0.0031	9.80E-07	0.0098	1.490	0.210	0.310	"
Soil, dry	1.09	0.0026	3.70E-07	0.0037	1.500	0.465	0.700	"
Sand	0.59	0.0014	4.70E-07	0.0047	1.520	0.190	0.290	"
Soil, wet	1.80	0.0043	4.50E-07	0.0045	1.810	0.525	0.950	"
Granite	2.89	0.0069	1.27E-06	0.0127	2.700	0.202	0.540	"
Average of composite materials:	*1.37*	*0.0033*	*6.13E-07*	*0.0061*	*1.670*	*0.432*	*0.632*	

(1) Andrews and Rodvey (1980)
(2) Geiger (1965)
(3) Nakshabandi and Kohnke (1965)
(4) Chow (1964) and Carslaw and Jaeger (1959)
(5) Hutchinson 1957, Jobson 1977, and Likens and Johnson 1969
(6) Cengel, Grigull, Mills, Bejan, Kreith and Bohn

Table 4-1. Thermal properties table taken directly from Pelletier and Chapra (2008).

4.3.5 Sensitivity Analysis/Parameter Estimation

Su (2019) stated that the Jordan River WASP model was developed to address several objectives. The second of these was to "Assess the water quality performance of the Jordan River over time, primarily over the summer months under elevated water temperatures and low flows." Given the nature of the water quality problems in the Jordan River, this should be a key focus of the WASP modeling.

The calibration effort conducted by the University of Utah used the WASP model to assess the Jordan River in light of climatic effects and population growth characteristics (Su 2020). Based on the documentation provided, the calibration did not include results for summer months or periods of low flow. Federal regulations show the Phase 2 TMDL must address critical conditions that lead to low DO in the lower Jordan River. Therefore, model calibration should be revisited and focus on 1) storm periods during times of high OM loading to ensure the appropriate accumulation of OM and 2) the summer low flow periods, since this is the critical time for high temperature and low DO. The availability of existing data to calibrate the model during the summer season is currently being evaluated by DWQ, and additional data collection may occur if necessary.

Once some of the important checks on flow volumes, routing, temperature, and inflow concentrations are completed, the sensitivity analysis should be revisited with appropriate parameters ranges. Given the number of unknown parameters, an automatic calibration algorithm would also be useful in determining appropriate values for the sensitive parameters. Hobson et al. (2014) documented ranges for the parameters within QUAL2K and QUAL2Kw. A large fraction of these are transferrable to WASP because many of the mechanisms represented in each model are similar.

4.4 APPLICABILITY OF JORDAN RIVER WASP APPLICATION FOR FUTURE LOAD ALLOCATIONS

The WASP model includes many of the features and mechanisms identified in prior studies as necessary to establish future load allocations in the Jordan River. While future efforts can detail the linkages between controlling processes and model capabilities, two model shortcomings may become important to future Jordan River planning. First, there appears to be a disconnect within the model between sediment transport (deposition and scour), organic matter (transport and deposition), and the sediment diagenesis module. This may limit understanding of dynamic sediment oxygen demands over space and time. Second, the model provides five different DO matter pools and only one particulate OM pool. Within the context of the Jordan River impairments, it may be necessary to include additional particulate OM pools (e.g., fine and coarse).

The rest of this section focuses on the current application of the Jordan River WASP model and suggests additional steps necessary to ensure that the model application is set up and calibrated in a way that maximizes model certainty in the complicated application of a time variable model.

For this modeling tool to be applied in the context of the Jordan River TMDL and the future work of determining load allocations, a summary of the suggestions is provided (see sections above for more detail):

1. Ensure flow volume and routing throughout the model domain are acceptable. Without correct water volumes and travel times, any water quality predictions will be inaccurate.
2. Determine if temperature predictions are reasonable throughout the model domain using available high frequency data. This step can help establish if inflows are represented well and if channel characteristics are reasonable.
3. Determine a more representative method for establishing reasonable point and nonpoint source concentrations based on the sparse data available. During this and the previous steps, specific conductance data that are typically included in high frequency sonde data can also be used to help determine how best to handle inflow volumes and concentrations.
4. Revisit the sensitivity analysis and calibration ensuring reasonable parameter ranges and an automatic calibration algorithm.

These efforts should focus on times when there are significant OM loads and the response to these loads during low flow periods.

5.0   CONCLUSIONS

Our understanding of OM in the LJR has improved because of recent research. Key points from the research reviewed in this synthesis are discussed below in regard to information that met the primary objectives for reviewing literature in this synthesis and the needs for information in the Phase 2 TMDL.

5.1 OXYGEN DEMAND

Recent research has provided new insights into oxygen demand in the water column and sediment. Key results that can be used to guide analysis in the Phase 2 TMDL are organized below according to important categories related to oxygen demand.

5.1.1 Algae Photosynthesis and Respiration

- Light measurements indicate sufficient light is available at depth to support benthic algae growth in the LJR except for several downstream sites (300 North, Cudahy Lane, SDSWRF and Burnham Dam). Results of water column light measurements contribute to understanding of controls on benthic algae growth in the LJR and potential contributions to SOD. Despite sufficient light and nutrients, measurements do not show additional growth of benthic algae in the LJR. This is due in part to erratic flows and unstable/smothered substrate that occurred during the survey. Seasonal Chl a measurements recorded below 7800 South indicate some influence of nutrient concentrations although no clear patterns are evident for some locations. Annual average calculations of Chl a along the length of the Jordan River indicate little variability. A closer look using monthly values may indicate temporal and spatial patterns that cannot be determined from annual averages. Results of this analysis could help identify parameters of concern for the Phase 2 TMDL.

- Typical diel DO cycles at some locations in the LJR indicate that respiration has a greater influence on the daily DO balance than photosynthesis or reaeration. Comparing diel DO at nine locations in the LJR shows an odd shift in peak DO with distance downstream, including three locations where peak DO occurs outside of the photo period. These results suggest that primary productivity in the UJR sets the minimum daily DO concentration entering the LJR, which is then lowered through decomposition of OM.   These results help define a spatial and temporal linkage between productivity in the LJR and sources that influence productivity.

5.1.2 Aerobic Decomposition in the Water Column

- Analysis of nutrient limitations on microbial growth suggests the Jordan River is more limited with respect to N than P, although the high LAP activity rates more likely reflect microbial utilization of the abundant N substrate. Observed variations in P acquisition are more variable and likely a response to available forms of organic C and N.

- The largest contribution of OM in the LJR is in the form of DOM which fuels net heterotrophy and metabolic consumption of DO. Future consideration of low DO during dry weather periods should identify factors that influence production of protein-rich DOM substrates. These results identify a characteristic of DOM that can influence chronic DO impairment and a need to identify sources of this type of DOM. This information would help with selecting a parameter of concern for the Phase 2 TMDL.

- Stable isotopes used to identify N transformation in the Jordan River indicate that denitrification does not have a strong impact and nitrification may be favored. This conclusion is speculative because the authors did not consider nitrate inputs with varying isotopic composition, and the conclusion runs counter to sediment N fluxes measured by Goel and Abedin (2016).

5.1.3 Macroinvertebrate Influence on Water Quality

- Based on literature values and estimated densities of invasive macroinvertebrate species *Corbicula* and *Potamopyrgu*s, large volumes of the Jordan River could be filtered on a daily basis.

- *Corbicula* communities generate oxygen demand on the order of about 1.7-2.5 percent of sediment oxygen demand. The net benefit of *Corbicula* and *Potamopyrgus* is speculative and would require 1) reduction of OM in the water column and sediment through filtering followed by 2) reduced turbidity and growth of benthic algae accompanied by photosynthetic oxygen production.

5.1.4 Hydrology and Oxygen Demand

- Isotope measurements and Bayesian model estimates indicate that dominant water sources to the Jordan River vary spatially and temporally over an annual cycle. Source water inputs (Utah Lake and groundwater) were evident in the LJR despite having low initial inputs compared to other sources. Cumulative flow from all three WRFs in the study reach remained consistent in the spring and fall (about 3.4 m^3/sec) but based on change input from other sources, total percent contribution from effluent was 19 percent in spring and 63 percent in fall. These results identify seasonal influences of pollutant loading in the LJR by pollutant source. Combined with other information, these results could help with load allocations in the Phase 2 TMDL.

- Dry weather flow increases to the LJR at 2100 South show a varied response in DO measurements depending on time of year and amount of flow change. Two flow change events in July and August 2015 showed an average increase of 0.32 mg/L DO at monitored sites 72 hours after the flow increase. Although increased flows have potential to increase DO and mitigate acute impairment, Phase 2 TMDL requirements will focus on load reductions to raise DO concentration and restore chronic impairment.

5.1.5 Sediment Oxygen Demand and Nutrient Processes

- Measured SOD rates in July and September 2015 from 1300 South and the Legacy Nature Preserve ranged 2.4–2.9 g-DO/m^2-d and were consistent with other SOD measurements collected in the LJR (Hogsett 2015).

- Direct measurements of sediment nutrient flux indicated dominance of denitrification over nitrification processes in the LJR sediments. These results identify a dominant biological process in the N cycle for the Jordan River and high potential in the river for removal of N.

- Microbial activity in sediments is limited by C during some seasons and by P during certain times of the year. These results provide some information on factors limiting microbial activity and follow traditional paradigms in comparison to the EEA ratios and patterns observed in the water column.

- Macroinvertebrate Influence on SOD

- Based on estimated densities in the Jordan River, *Corbicula* can consume significant amounts of N and P at levels that contribute to SOD in the LJR. However, rates of excretion are reported at twice the level of assimilation rates which is biologically impossible. Additional information is needed to clarify these results.

- Mean DO consumption and CO_2 respiration rates from *Corbicula* in Jordan River run habitat are estimated at 2.10 mg/m^2-hr and 1.0 mg/m^2-hr, respectively. At this rate of DO consumption, Corbicula would contribute 2.5 percent of SOD measured at 2 g/m^2-d. This contribution was measured as part of field SOD measurements collected by Hogsett (2017).

5.2 TOTAL ORGANIC MATTER AND ORGANIC MATTER SOURCES

Research of TOM loading in the LJR and OM sources have identified new information that will be needed for the Phase 2 TMDL. Key points identified in this section are summarized below. They include the following:

- CPOM loading from major Jordan River tributaries (i.e., BCC, Little Cottonwood Creek, and Mill Creek) is estimated at 200,000 kg/yr during normal to high years and less than half this amount during low runoff years. Monthly calculations of CPOM loads in the LJR indicate deposition in some segments during some years that could contribute to SOD. Individual data records are needed to verify the conclusions made in Miller (2019c).

- Measurements collected by Miller (2019c or d) are approximately 2–5 times larger than those reported by Epstein et al. (2016). Measurements of CPOM samples collected during both research studies have potential for high variability among samples, sampling methods, and over time. Review of original data records (including laboratory results and calculations) and additional analysis is needed to explain these differences. This information is necessary to determine pollutant sources that contribute to measured SOD levels in the LJR.

- Review of monthly TSS and VSS measurements have not identified significant spatial or temporal patterns. However, original data records are needed to make meaningful comparisons between individual sites and years. Results of more detailed analysis would help with selecting a parameter of concern for the Phase 2 TMDL.

- Based on FI values developed from EEM analysis, microbes may comprise a significant portion of DOM in the water column. EEMs have not been widely used in urban rivers to analyze DOM. Urban rivers could include constituents that modify FI values reported from other environments. These results indicate that microbe sources and resources that influence microbe activity could have a significant role in water column oxygen demand and decomposition of OM.

- FI values increase downstream of WRF effluent discharges during all seasons, which suggests that treatment plant effluent influences OM composition in the Jordan River. FI values indicate high levels of DOM in the fall and a limited influence by natural terrestrial sources (e.g. leaves, litter). These results suggest that the presence of aquatic DOM and nutrients could contribute to chronic low DO levels in the fall season.

- Ecoenzyme measurements indicate the C included in Jordan River OM is labile, including fractions of dissolved and particulate OM. Labile OM is decomposed more rapidly than recalcitrant OM, creating a relatively greater contribution to oxygen demand.

- Analysis of stable isotopes indicates Jordan River CPOM consists primarily of terrestrial sources except during summer months when macrophytes contribute equal amounts. FPOM is comprised primarily of terrestrial and WRF sources in the fall and BOM and Utah lake in the summer. The primary source of Jordan River DOM is from Utah Lake with major contributions from WRFs throughout the year. However, the methodology of this assessment does not consider periodic contributions from stormwater. These results provide significant contributions to understanding OM pollutant sources and potential contributions to SOD in the LJR based on particle size.

- Although FPOM isotope signatures could not be correlated to effluent discharge, values downstream of CVWRF were consistently high in the river indicating a consistent influence on FPOM signatures at this location. These results may be related to the treatment design at the facility as this pattern was not observed at other WRFs.

- Paired measurements of fDOM loading from the 1300 South stormwater outfall and Jordan River at 1700 South and fDOM:BODu relationships indicate BODu loading from stormwater can exceed

1,200 percent of the Jordan River background load when the relative degradability of DOM is accounted for. On average, stormwater from 1300 South accounts for >26 percent of the mainstem Jordan River BODu load, with routine excursions >100 percent even during small storms, indicating that stormwater is a significant and on-going intermittent source of oxygen demand contributing to both acute and chronic DO impairment. Combined with previous work from Richardson (2014) these results contribute significant understanding to the labile nature of OM in stormwater and pollutant source loading.

- *Corbicula* function as a source and a sink of OM in Jordan River sediments.

- Impacts of climate change are projected to move the 50th percentile of mean annual stream flow in the Jordan River and tributaries by 4 weeks in the 2040s, and 8 weeks in the 2090s. Flow increases in the 2040s and 2090s are predicted to range from 7–11 percent for BCC and 14–15 percent for the Jordan River, respectively. The respective increases in sediment load in the 2040s and 2090s are projected to range from 7–40 percent in the canyons and about 7–14 percent in the Jordan River Valley. The Phase 2 TMDL will need to account for changes in future loading. These results will help meet that requirement.

- Based on past success, Jordan River restoration has the potential to restore features that could reduce pollutant loading in support of beneficial uses. A critical aspect to restoration in the LJR will be increasing baseflow conditions and promoting a more natural hydrograph. These findings indicate the potential to remove accumulated OM and potentially reduce SOD through flow management.

5.3 JORDAN RIVER WASP IMPLEMENTATION

Recent modeling efforts have produced a calibrated dynamic water quality and flow model of the Jordan River. A summary of key results applicable to the LJR river segment include:

- The WASP model appears to slightly overpredict the DO concentration in the LJR as compared to measured data.

- Overprediction of DO in the LJR may be due to simulated SOD values produced by the sediment diagenesis routine. This may be due to simulated SOD values which are consistently lower than those measured by Hogsett (2015).

- Modeled DO concentrations are heavily influenced by the model input parameter O_2:C production by macroalgae. The observed dependence of DO on macro/benthic algae could be another reason for overpredicting DO, which suggests there are underlaying processes that appear to not be captured by the WASP model as implemented.

- General agreement is apparent between measured values and modeled estimates of CBOD/BODu, phytoplankton/chlorophyll-a, and total phosphate.

- Modeled sediment-ammonia benthic flux produced by the WASP sediment diagenesis routine is substantially lower than measurements by Hogsett (2015), suggesting that some processes influencing flux may not be captured by the WASP model.

- Except for a few outliers in some segments, model estimates of phytoplankton/chlorophyll-a agree with measured values.

- Both ammonia (as NH_3-N) and nitrite-nitrate (NO_2-NO_3-N) are underpredicted in water column, as is DON below 5400 South.

5.4 SUMMARY OF RESULTS – SYNTHESIS 1 AND 2

The 2017 synthesis of Jordan River research reviewed research and monitoring projects completed 2010–2015 that helped to fill data gaps identified as the Phase 1 TMDL was completed. Results in that synthesis characterized OM pollutant sources, identified linkages between water-quality impairment and OM pollution, and further defined DO impairment in the LJR.

Some results described in this current synthesis contribute to understanding of similar topics addressed in the first synthesis. Other results help to define a parameter of concern linking OM and DO demand with pollutant sources, OM lability, and OM contributions to SOD.

This section summarizes results of each synthesis and identifies how this information can be used in the Phase 2 TMDL. Table 5-1 organizes the results (referenced to their source) according to the important processes and linkages displayed in Figure 1-1 including DO demand, DO demand in the water column and sediment, OM characteristics (e.g., particle size, lability, etc.), OM pollutant sources, and influences on OM.

Table 5-1. Conclusions regarding important processes and linkages and implications for TMDL.			
Process/Issue/Concern	**Synthesis 1 (2017)**	**Synthesis 2 (2020)**	**Implications for Phase 2 TMDL**
Dissolved Oxygen Demand			
Stream Metabolism	UJR is net autotrophic; LJR is net heterotrophic.[1,2,3] UJR has high primary production in water column and benthos.[11]	Sufficient light exists to support benthic algae growth in LJR, but benthos is limited in spite of high nutrients.[15] Peak DO in the LJR outside of photoperiod indicates minimum DO levels in critical season are set by primary production in the UJR.[16] Net heterotrophy in the LJR is fueled by DOM and metabolic consumption of DO.[17]	Limited productivity in LJR. Algal productivity in UJR contributes OM loads to LJR. OM sources in the UJR significantly influence diel DO in the LJR. Factors that influence production of protein-rich DOM (including algae contributions) must be considered to mitigate low DO during dry conditions. Data is needed to distinguish between DOM sources (e.g., WRF effluent or algae).
Hydrology and Oxygen Demand	Measured Jordan River reaeration rates are greatest in middle sections of the UJR and relatively less near Utah Lake and the LJR due to lower channel gradients.[35] 2100 South diversion to LJR increases DO through reaeration by 0.3 mg/L.[36]	Dry weather flow contributions at 2100 South during summer months can increase DO from upstream contributions and increased reaeration.[34]	Opportunities for improving DO levels should be considered as potential improvements to chronic DO impairment.

Table 5-1 (cont'd). Conclusions regarding important processes and linkages and implications for TMDL.

Process/Issue/Concern	Synthesis 1 (2017)	Synthesis 2 (2020)	Implications for Phase 2 TMDL
SOD and Nutrient Processes	SOD accounts for 54% of ambient DO deficit in LJR.[5] SOD is influenced by spring runoff[6], and levels increase during the winter.[7] CPOM contributing to SOD is primarily leached of OM.[4] Methane production contributes 56% of total SOD.[8]	Denitrification appears to dominate N transformation processes in the Jordan River based on measured sediment N flux.[18] Sediment microbial activity is limited seasonally by C and P.[19] Macroinvertebrates function as a sink of DO in sediment.[39]	CPOM contributing to SOD supports chronic impairment. Episodic release of methane gas in the LJR may contribute to acute impairment. Sediment microbial activity can be managed by controlling nutrient loads. Net influence of macroinvertebrates on DO demand is part of SOD measurements.
Total Organic Matter and Organic Matter Sources			
OM Particle Size	Annual OM budget shows DOM is major component (91%) of total annual load followed by FPOM (8%), then CPOM (1%).[9]	CPOM monitoring results are 2–5 times larger than OM budget measurements (2010–2014).[25] Peak monthly loads typically occur in spring of each year.[26] Some measurements indicate CPOM settling in LJR.[27] Monthly measurements of FPOM as VSS (2009–2012) show both inconsistency and general trends at monitoring sites.[28]	DOC and VSS should be considered for parameters of concern. CPOM measurement methods have potential for high variability among samples, sampling methods, and over time. Load allocations should account for this variability. Original VSS monitoring records are needed for further analysis of FPOM. Quantify CPOM deposition and corresponding O consumption, compare to SOD measurements.
OM Characteristics	Organic compounds are rapidly leached (80–90%) from terrestrial OM within 3 hours.[4] Decomposition of sediment OM under aerobic conditions takes approximately 1 year and under anaerobic conditions takes about 3–5 years.[24]	Dissolved and particulate forms of Jordan River OM supporting microbial activity are labile.[21, 36] Microbes may comprise a significant portion of DOM in Jordan River.[38] WRF effluent influences OM composition in the Jordan River.[23]	Water column measurements of OM generally indicate contributions to acute impairment while sediment OM indicates contributions to chronic impairment. Identify urban constituents that could potentially modify FI measurements of DOM. Link WRF effluent to OM composition and resulting O demand.

Table 5-1 (cont'd). Conclusions regarding important processes and linkages and implications for TMDL.

Process/Issue/Concern	Synthesis 1 (2017)	Synthesis 2 (2020)	Implications for Phase 2 TMDL
OM Sources	UJR is a significant source of OM due to eutrophication and could account for 55 percent of OM loads to LJR.[10] DOM and CPOM are primarily from terrestrial sources, and FPOM is from terrestrial and aquatic sources.[12] C:N ratios for FPOM are similar to aquatic sources; ratios for CPOM are similar to terrestrial sources.[13]	Summer loads of CPOM equally consist of aquatic (macrophytes) and terrestrial OM. FPOM is primarily terrestrial source (i.e., leaves, litter, WRF effluent) during fall and aquatic source (i.e., river benthos, Utah lake) during summer. DOM is primarily Utah Lake with major contributions from WWTPs throughout the year.[31] LJR flow contributions vary seasonally with WRF effluent dominating (63%) in the fall.[29] Monitoring sites downstream of WWTP effluent had OM with significant WWTP characteristics including microbial derived, protein-like DOM.[30] *Corbicula* biologically function as a source and sink of OM in sediment.[22]	Results should be used to consider seasonal load allocations depending on load contributions during critical period when chronic impairment exists. Stormwater OM should be accounted for during dry and wet (storm event) weather. Research results are needed to distinguish between WRF effluent and algae.
Stormwater OM	Episodic releases of highly degradable DOM are related to sudden DO depletions in LJR.[14]	Stormwater discharge from 1300 South can account for 26% to more than 100% of BODu in the Jordan River even during small storms. Loading can exceed 1,200% when degradability of DOM is considered.[20]	Stormwater loading contributes substantially to acute impairment in the LJR. Partially degraded stormwater OM could contribute to chronic impairment in the LJR.
Future OM Changes		Climate change predictions estimate median flow occurring 1 month earlier in 2040 and 2 months earlier in 2090. Tributary and Jordan River flows would increase by 11% and 15%, respectively by 2090. Sediment loads would increase by 40% in	Use these results to account for portion of TDML reserved for future growth.

Table 5-1 (cont'd). Conclusions regarding important processes and linkages and implications for TMDL.

Process/Issue/Concern	Synthesis 1 (2017)	Synthesis 2 (2020)	Implications for Phase 2 TMDL
		tributaries and 14% in the Jordan River during that same time period.[32]	
Water Quality Modeling			
Jordan River WASP	Increased base flow (25%) could improve DO above 5.5 mg/L chronic criteria in LJR.[37]	Acceptable WASP model results for estimating BODu, phytoplankton, Chl a, and total phosphate. WASP model overestimates DO in the LJR and underestimates sediment-ammonia flux and water column ammonia, nitrite-nitrate, and DON.[33]	WASP model is structurally sound but improvements are needed to link sources with water quality in support of load allocations.

[1] DWQ 2015a p. 22
[2] Epstein et al. 2015 p. 17
[3] Hogsett 2015 p. 142
[4] Richardson 2014 p. 37
[5] Hogsett 2015 pp. 125-126
[6] Hogsett 2015 pp. 119-122
[7] Hogsett 2015 pp. 128-130
[8] Hogsett 2015 pp. 185-186
[9] Epstein et al. 2015 pp. 15, 20-21
[10] Hogsett 2015 p. 205

[11] Hogsett 2015 p. 203
[12] Kelso 2015 pp.2-3
[13] Epstein et.al 2015 p.18
[14] Richardson 2014 pp. 41-43
[15] Miller 2019a Table 3
[16] Cirrus 2017b Figure 3
[17] Follstad Shah et.al. 2019b p.20
[18] Goel and Abedin 2016 p.11
[19] Follstad Shah 2017 p.11
[20] Dupont 2019 p.10

[21] Follstad Shah et al. 2019b p. 19
[22] Richards 2018 pp. 278-282
[23] Follstad Shah et al. 2017 Figure 10
[24] Hogsett 2015 pp. 166-167, 186
[25] Miller 2019c p. 123
[26] Miller 2019c pp. 118-122
[27] Miller 2019c p. 115
[28] Miller 2019b pp. 134-143
[29] Follstad Shah 2019a p. 11
[30] Kelso 2018 pp. 32-34, 65-66, 83-86

[31] Kelso 2018 pp. 28-38
[32] Khatri et al. 2019 pp.1559-1560
[33] Su 2019 pp. 31-33
[34] Cirrus 2016 Table 3-4
[35] Hogsett 2015 Table 2
36: Salt Lake City 2015 p. 7
[37] SWCA 2013 Figure 8
[38] Follstad Shah et al. 2017 p. 10
[39] Richards 2018 pp. 291

5.5 DATA GAPS

Some data gaps currently remain in our understanding of DO demand in the LJR. DWQ will consider these gaps and determine which should be addressed prior to or during the Phase 2 TMDL.

5.5.1 Oxygen Demand

- Oxygen demand in the water column and sediment should be defined as part of an oxygen budget for the LJR. Variability of these processes should be defined by season, wet and dry years, and other scenarios that capture external and internal influences on DO demand. Several studies have identified SOD by segment (assuming similar substrate and consistent coverage across the full width and length of river segments) and by winter and summer seasons. Most current research does not explicitly address water column demand.

- Past and current measurements of SOD at specific locations in the LJR indicate this process generates the most oxygen demand in the LJR. More information would be helpful to quantify

spatial variability of SOD regarding changes in substrate and sediment OM content in the LJR before these measurements can be extrapolated to the scale of river segments.

- Recent research has indicated that although microbial communities respond to nutrient inputs, Jordan River microbes do not appear to be growth limited based on traditional assessment methods. Additional research and analysis are needed to determine if microbial activity in the water column and sediment can be regulated through pollutant source management, particularly regarding protein-rich DOM during periods of low DO in low flow periods.

- Research is needed to identify what factors are suppressing and modifying primary production in the LJR at sites where peak DO concentration occurs outside the photoperiod.

- Additional analysis is needed to compare phytoplankton concentration with benthic algae biomass to determine dominant drivers of photosynthetic production in the UJR and LJR.

- Research is needed to measure existing densities of *Corbicula* and *Potamopyrgus* in segments of the Jordan River that are not currently surveyed (Richards 2018). Research should include in situ measurements of density as well as current methods for estimating rates of C, N, and P filtering, consumption, and excretion, as well as carbon fixation (Same and Olenin 2005). Findings should be analyzed to determine the influence of benthic invertebrates on Jordan River water chemistry and considered in future Jordan River water quality assessments.

5.5.2 Total OM and OM pollutant sources

- Future OM load increases and timing should be calculated based on estimates of future changes to flow and sediment delivery. This effort must capture or attempt to capture the changes expected in urban stormwater runoff. Research indicates that oxygen demand discharged by stormwater outflow completely overwhelms upstream conditions. Without evaluating future change to this source, fundamental contributions will go unrecognized and support for improving stormwater management in drainage areas will be lost.

- A copy of the original data records and statistical analysis of water quality monitoring completed by Miller (2019a, 2019b, and 2019c) should be obtained and used to complete a seasonal analysis to identify temporal patterns and significant differences.

- How OM loading contributes to SOD (i.e., how does SOD accumulate) should be quantified, and a method to allocate contributions between pollutant sources should be identified.

- Additional OM budget assessments should be completed to identify particle size contributions during a wet year.

- Oxygen demand by season generated by microbial consumption of DOM should be quantified.

- The accuracy of EEM analysis in urban streams settings and potential sources of interference should be validated.

- Stream channel restoration to increase the LJR channel gradient between 2100 South and the PacifiCorp diversion should be investigated as a measure to increase reaeration and DO concentrations to downstream LJR segments.

- Modification of the 2100 South diversion structure should be investigated as a way to reduce OM loading to the LJR.

5.5.3 Jordan River WASP Implementation

- Additional investigations are needed to adjust model input parameters that simulate the sediment diagenesis routines to increase modeled SOD values, particularly downstream of the Surplus Canal.

- It is important to ensure flow volume and routing throughout the model domain are acceptable. Without correct water volumes and travel times, any water quality predictions will be inaccurate.

- Available high frequency data should be used to determine if temperature predictions are reasonable throughout the model domain. This step will help establish if inflows are represented well and if channel characteristics are reasonable.

- A more representative method is needed for establishing reasonable point and nonpoint source concentrations based on the sparse data available. During this and the previous steps, specific conductance data that are typically included in high frequency sonde data can also be used to help determine how best to handle inflow volumes and concentrations.

- The sensitivity analysis and calibration should be revisited to ensure reasonable parameter ranges and establish an automatic calibration algorithm.

REFERENCES

Aqua Engineering Inc. 2013. Jordan River Water Quality Aeration Enhancement Project. Draft project report to Salt Lake City Corporation. September 13, 2013.

Borup, B. and S. Smith. 1999. An Assessment of Water Quality in the Jordan River from 1990 to 1997. Civil and Environmental Engineering Department, Brigham Young University, Provo, Utah. Prepared for the State of Utah, Department of Environmental Quality, Division of Water Quality.

Bott, T.L. 2007. Primary productivity and community respiration. In: Hauer Fr, Lamberti GA (eds) Methods in stream ecology. Academic Press, San Diego, p. 68.

Butts, T. A., and R.L. Evans. 1978. Sediment oxygen demand studies of selected northeastern Illinois streams. Illinois State Water Survey, Circular 129, NTIS Publication No. PB-285-788. Springfield, VA: NTIS.

Butts, T. A., and R.L. Evans. (1978). "*Sediment oxygen demand studies of selected northeastern Illinois streams*." Illinois State Water Survey, Circular 129, NTIS Publication No. PB-285-788. Springfield, VA: NTIS.

Burns, R.G. 1978. Soil Enzymes. Academic Press. New York, London.

Cirrus (Cirrus Ecological Solutions, LC.). 2009. Jordan River TMDL: Public Draft Work Element 2 – Pollutant Identification and Loading. Prepared by Cirrus Ecological Solutions, LC and Stantec Consulting, Inc. for Utah Division of Water Quality. January 2009.

Cirrus. 2016. Lower Jordan River: Phase 2 Report – Dissolved Oxygen Response to Flow Change (2011-2016). Prepared by Cirrus Ecological Solutions, LC for Utah Division of Water Quality. March 2016.

Cirrus. 2017a. Jordan River Research 2010–2015. Prepared by Cirrus Ecological Solutions, LC for Utah Division of Water Quality. October 2017.

Cirrus. 2017b. Lower Jordan River Diurnal Dissolved Oxygen Assessment (July 29 – August 4, 2017). Prepared by Cirrus Ecological Solutions, LC for Jordan River Commission. December 2017.

Cole, T.M. and E.M. Buchak. 1994. "CE-QUAL-W2: A Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model, Version 2.0, User Manual." Draft Instruction Report, Station, Vicksburg, MS.

Cole, T.M. and S.A. Wells. 2003. "CE-QUAL-W2: A two-dimensional, laterally averaged, Hydrodynamic and Water Quality Model, Version 3.1." Instruction Report EL-03-1, US Army Engineering and Research Development Center, Vicksburg, MS.

DWQ (Utah Division of Water Quality). 2013. Jordan River Total Maximum Daily Load Water Quality Study – Phase 1. Prepared by Cirrus Ecological Solutions, LC and Stantec Consulting Inc. for Utah Division of Water Quality. July 2013.

DWQ. 2015a. Jordan River 2014 Synoptic Survey and Model Validation Report. Utah Division of Water Quality, December 2015.

DWQ. 2015b. Continuous DO and fDOM monitoring data (2014) collected from in-situ probes installed at four locations on the lower Jordan River. Utah Division of Water Quality.

DWQ. 2016. Utah's Final 2016 Integrated Report. Salt Lake City, Utah. Utah Department of Environmental Quality. Final 2016 IR: version 2.1. Available at: http://www.deq.utah.gov/ProgramsServices/programs/water/wqmanagement/assessment/currentIR2016.htm

Dupont, R.R., J. Richardson, and D. Willey. 2018. Demonstrating the Pollutant Loading from Stormwater Discharge to an Urban River in the Intermountain West Using High-Frequency Data. Manuscript of platform presentation at the 2018 StormCon Conference, Denver, CO, August 14, 2018.

EPA (U.S. Environmental Protection Agency). 2006. Guidance Clarification Regarding "Phased" Total Maximum Daily Loads. Memorandum to Water Division Directors Regions I-X, August 2, 2006. Available   from http://www.epa.gov/OWOW/tmdl/decisions/http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/tmdl_clarification_letter.cfm

Epstein, D., J. Kelso, and M. Baker. 2016. Beyond the urban stream syndrome: organic matter budget for diagnostics and restoration of an impaired river. Urban Ecosystems (2016) 19:1623-1643. DOI 10.1007/s11252-106-0556-y.

Follstad Shah, J., R. Smith, R. Gabor, S. Weintraub, Y. Jameel, and M. Navidomskis. 2017. Nitrogen sources and transformations within the Jordan River, Utah and Microbial community response to energy and nutrient availability in the Jordan River, Utah. Progress report to the Jordan River Farmington Bay Water Quality Council. Pp. 26. December 21.

Follstad Shah, J., R. Smith, R. Gabor, Y. Jameel, and S. Weintraub. 2018. Linking Water Sources & Water Quality within the Jordan River, Utah. Salt Lake County Watershed Symposium. Pp. 23. November 15.

Follstad Shah, J., Y. Jameel, R.M. Smith, R.S. Gabor, P.D. Brooks, and S.R. Weintraub. 2019a. Spatiotemporal Variability in Water Sources Controls Chemical and Physical Properties of a Semi-arid Urban River System. *Journal of the American Water Resources Association* 1-17. https://doi.org/10.1111/1752-1688.12734.

Follstad Shah, J., S. Weintraub, R. Smith, R. Gabor, and Y. Jameel. 2019b. Microbial Community Response to Shifting Water Quality and Quantity in an Arid Urban Ecosystem. Society of Freshwater Science Annual Meetings. Pp. 23. May 21.

Follstad Shah, J. 2020. Personal communication between Dr. Jennifer Follstad Shah (University of Utah) and Lucy Parham (Utah Division of Water Quality).

Goel, R. and M. Hogsett. 2009. Interim Status Report [on Sediment Oxidation Demand and Reaeration]. University of Utah. Salt Lake City, Utah.

Goel, R. and M. Hogsett. 2010. SOD, nutrient fluxes, sediment mineralogy and reaeration in the Jordan River. presentation at 4th Salt Lake Countywide Watershed Symposium. University of Utah.

Goel, R. and S. Abedin. 2016. Nitrogen dynamics in the Jordan River and Great Salt Lake wetlands. Research status report prepared by R. Goal and S. Abedin for Jordan River Farmington Bay Water Quality Group. May 2016.

Hakenkamp, C. and M. Palmer. 1999. Introduced bivalves in freshwater ecosystems; the impact of Corbicula on organic matter dynamics in a sandy stream. Oecologia. 119: 445-451.

Hobson, A.J., B.T. Neilson, N. von Stackelberg, M. Shupryt, J. Ostermiller, G. Pelletier, and S.C. Chapra. 2014. "Development of minimalistic data collection strategy for QUAL2Kw." *Journal of Water Resources Management and Planning*. https://doi.org/10.1061/(ASCE)WR.1943-5452.0000488

Hogsett, M. 2015. Water Quality and Sediment Biogeochemistry in the Urban Jordan River, UT. Dissertation submitted to Department of Civil and Environmental Engineering, University of Utah. May 2015.

Hogsett, M., and R. Goel. 2013. "Dissolved Oxygen Dynamics at the Sediment–Water Column Interface in an Urbanized Stream." *Environmental Engineering Science*, *30*(10), 594-605.

Jensen, S.F. 1994. Section 305b Water Quality Assessment Update for the Jordan River Sub-Basin. Board of Salt Lake County Commissioners, Commission Staff Office. February 1994. 48 pages.

Kelso, J. 2018. Organic Matter Sources, Composition, and Quality in Rivers and Experimental Streams. Dissertation submitted to Utah State University, Logan Utah.

Kelso, J. and M. Baker 2017. FPOM and DOM isotope values, HydroShare, http://www.hydroshare.org/resource/4eb5c9c871e34aa4ae6951ce6d15020d

Khatri, K.B., C. Strong, N. von Stackelberg, M. Buchert, and A.K. Kochanski. 2019. Impact of Climate and Land Use Chand on Streamflow and Sediment Yield in a Snow-dominated Semiarid Mountainous Watershed. *Journal of the American Water Resources Association* 54 (4): 1540-1563. https://doi.org/10.1111/1752-1688.12803.

Miller, T. 2019a. Chapter 2: Light Attenuation and Nutrients in the Jordan River and Their Relationship to Periphyton and Phytoplankton Communities. Wasatch Water Quality Council. Pp. 67. August.

Miller, T. 2019b. Chapter 4: Diel Patterns of Dissolved Oxygen, Flourescing Dissolved Organic Matter and Turbidity and Their Relationship to Seasonal Runoff and Storm Events in the Jordan River. Wasatch Front Water Quality Council. Pp. 52. August.

Miller, T. 2019c. Chapter 3: Course Particulate Organic Matter Measurements in the Jordan River and its Major Tributaries, 2010-2015. Wasatch Front Water Quality Council. Pp. 38. August.

Neilson, B.T. 2011. "Two-zone Dynamic Temperature Model Development and Calibration for the Jordan River." Project Report. Utah Department of Environmental Quality, Division of Water Quality. Salt Lake City, UT.

Overbeck J. 1991. Early studies on ecto- and extracellular enzymes in aquatic environments *In*: Chrost. 1991. Microbial Enzymes in Aquatic Environments. New York: Springer-Verlag, pp. 1–5

Pelletier, G. and S.C. Chapra. 2008. QUAL2Kw theory and documentation (version 5.1): A modeling framework for simulating river and stream water quality. Environmental Assessment Program. Olympia, Washington.

Richards, D.C. 2017. Lower Mill Creek and Mid-Jordan River Native Mussel Survey 2017. As it pertains to the Central Valley Water Reclamation Facility Discharge. Report to: Wasatch Front Water Quality Council. Salt Lake City, UT.

Richards, D.C. 2018. Chapter 12: A Snail, a Clam, and the River Jordan: A Revealing Novel. Wasatch Front Water Quality Council. Pp. 49. March 1.

Rushforth, S. and S. Rushforth. 2009. Assessment of the Phytoplankton Floras of the Jordan River June – October 2009. December 2009.

Salt Lake City. 2015. Monitoring Results of DO Profiling and Water Quality Sampling during Storm Water Events and Dredging Activities. Investigative reports (November 2014–October 2015) prepared by SWCA Environmental Consultants for Salt Lake City Public Utilities.

Salt Lake County Watershed Planning and Restoration Program. 2018. Section 319 Nonpoint Source Pollution Control Program Watershed Project Final Report. Pp. 42.

Sinsabaugh, R.L. and J.J. Follstad Shah. 2012. Ecoenzymatic Stoichiometry and Ecological Theory. Annual Review of Ecology, Evolution, and Systematics. 2012.43:313-343. Downloaded from www.annualreviews.org May 7, 2020.

Stephens, D.W. 1984. Dissolved Oxygen Regime of the Jordan River, Salt Lake County, Utah. U.S. Geological Survey. Prepared in cooperation with the Salt Lake County Division of Flood Control and Water Quality, Salt Lake City, Utah. Water Resources Investigations Report 84-4056.

SWCA Environmental Consultants. 2014. Lower Jordan River: Plan for Flow Experiments (2014–2016). Prepared by SWCA Environmental Consultants for River Network. May 2014.

Thiros, S.A. 2003. Quality and Sources of Shallow Ground Water in Areas of Recent Residential Development in Salt Lake Valley, Salt Lake County, Utah. U.S. Geological Survey Water-Resources Investigations Report 03-4028. National Water-Quality Assessment Program.

Thompson, R. 2020. Section Manager Watershed Planning and Restoration, Salt Lake County, Utah. Personal communication with Eric Duffin, Watershed Scientist/Hydrologist, Cirrus Ecological Solutions, LC, Logan, Utah.

USGS (United States Geological Survey). 1985. Methods for the Determination of Inorganic Substances in Water and Fluvial Sediments, Techniques of Water-Resources Investigations of the United States Geological Survey, Book 5, Chapter A1 Edited by Marvin J. Fishman and Linda C. Friedman.

Utah Division of Water Quality. 2016. Utah's Final 2016 Integrated Report. Salt Lake City, Utah. Utah Department of Environmental Quality. Final 2016 IR: version 2.1. Available at: http://www.deq.utah.gov/ProgramsServices/programs/water/wqmanagement/assessment/currentIR2016.htm

Wool, T.A., R.B. Ambrose, and J.L. Martin. 2008. "WASP8 Temperature Model Theory and User's Guide: Supplement to Water Quality Analysis Simulation Program (WASP) User Documentation." US Environmental Protection Agency, Office of Research and Development. Washington, DC.

APPENDIX A: LITERATURE REVIEWED

Cirrus Ecological Solutions, LC. 2017. Lower Jordan River Diurnal Dissolved Oxygen Assessment (July 29 – August 4, 2017). Jordan River Commission. Pp. 9. December 11.

Dupont, R.R., J. Richardson, and D. Willey. 2018. Demonstrating the Pollutant Loading from Stormwater Discharge to an Urban River in the Intermountain West Using High-Frequency Data. Manuscript of platform presentation at the 2018 StormCon Conference, Denver, CO, August 14, 2018.

Follstad Shah, J., R. Smith, R. Gabor, S. Weintraub, Y. Jameel, and M. Navidomskis. 2017. Nitrogen sources and transformations within the Jordan River, Utah and Microbial community response to energy and nutrient availability in the Jordan River, Utah. Progress report to the Jordan River Farmington Bay Water Quality Council. Pp. 26. December 21.

Follstad Shah, J., R. Smith, R. Gabor, Y. Jameel, and S. Weintraub. 2018. Linking Water Sources & Water Quality within the Jordan River, Utah. Salt Lake County Watershed Symposium. Pp. 23. November 15.

Follstad Shah, J., S. Weintraub, R. Smith, R. Gabor, and Y. Jameel. 2019b. Microbial Community Response to Shifting Water Quality and Quantity in an Arid Urban Ecosystem. Society of Freshwater Science Annual Meetings. Pp. 23. May 21.

Follstad Shah, J.J., Y. Jameel, R.M. Smith, R.S. Gabor, P.D. Brooks, and S.R. Weintraub. 2019a. Spatiotemporal Variability in Water Sources Controls Chemical and Physical Properties of a Semi-arid Urban River System. *Journal of the American Water Resources Association* 1-17. https://doi.org/10.1111/1752-1688.12734.

Goel, R. and S. Abedin. 2016. Nitrogen Dynamics in the Jordan River and Great Salt Lake Wetlands. Department of Civil & Environmental Engineering, the University of Utah. Pp. 22. May.

Kelso, J. 2018. Organic Matter Sources, Composition, and Quality in Rivers and Experimental Streams. Dissertation submitted to Utah State University, Logan Utah.

Khatri, K.B., C. Strong, N. von Stackelberg, M. Buchert, and A.K. Kochanski. 2019. Impact of Climate and Land Use Chand on Streamflow and Sediment Yield in a Snow-dominated Semiarid Mountainous Watershed. *Journal of the American Water Resources Association* 54 (4): 1540-1563. https://doi.org/10.1111/1752-1688.12803.

Miller, T. 2019a. Chapter 2: Light Attenuation and Nutrients in the Jordan River and Their Relationship to Periphyton and Phytoplankton Communities. Wasatch Water Quality Council. Pp. 67. August.

Miller, T. 2019. Chapter 3: Course Particulate Organic Matter Measurements in the Jordan River and its Major Tributaries, 2010-2015. Wasatch Front Water Quality Council. Pp. 38. August.

Miller, T. 2019b. Chapter 4: Diel Patterns of Dissolved Oxygen, Flourescing Dissolved Organic Matter and Turbidity and Their Relationship to Seasonal Runoff and Storm Events in the Jordan River. Wasatch Front Water Quality Council. Pp. 52. August.

Miller, T.G. 2019. Flow, Temperature, pH and Ammonia in the Jordan River and State Canal with Reference to the 1999 and 2013 Ammonia Criteria and 2019 South Plant Notice of Violation. South Davis Sewer District. Pp. 30. January 4.

Richards, D.C. 2018. Chapter 12: A Snail, a Clam, and the River Jordan: A Revealing Novel. Wasatch Front Water Quality Council. Pp. 49. March 1.

Salt Lake County Watershed Planning and Restoration Program. 2018. Section 319 Nonpoint Source Pollution Control Program Watershed Project Final Report. Pp. 42.

Su, Juhn-Yuan. 2019. Jordan River Modeling Through the Water Quality Assessment Simulation Program (WASP): Model Report. Department of Civil and Environmental Engineering, University of Utah. Pp. 208. December 2.

SUPPORTING STUDIES

Richards, D.C. 2016. Comparison between two Macroinvertebrate Assessments in Lower Mill Creek. Jordan River/Farmington Bay Water Quality Council. Pp. 18. July 17.

Richards, D.C. 2016. Does Phylogeny Predict Sensitivity to Ammonia in Freshwater Animals using USEPA Ammonia Criteria Data? Jordan River Farmington Bay Water Quality Council. Pp. 36. April 28.

Richards, D.C. 2016. Jordan River Watershed Mollusk Surveys: "Last of the Anodonta". 2016 Annual Report. Jordan River/Farmington Bay Water Quality Council. Pp. 47. September 19.

Richards, D.C. 2016. Macroinvertebrate Assemblages Upstream and Downstream of Two Wastewater Treatment Facilities on Lower Jordan River and State Canal. Progress Report. South Davis Sewer District. Pp. 27. November 16.

Richards, D.C. 2016. Numeric Water Quality Criteria Recalculation for Several Toxicants Related to Central Valley Water Reclamation Facility Discharge into Mill Creek, Salt Lake County, UT. Part 1: Ammonia. Technical Report. Jordan River/Farmington Bay Water Quality Council. Pp. 69. April 15.

Richards, D.C. 2016. Real and Perceived Macroinvertebrate Assemblage Variability in the Jordan River, Utah can Affect Water Quality Assessments. Draft Technical Report. Pp. 29. August 16.

Richards, D.C. 2018. Concept Reference, Conservation Status, Distribution, Ecology and Life History, Economic Attributes, Management Summary, Population/Occurrence Viability, Authors/Contributors, and References for fish species potentially occurring in Lower Jordan River and State Canal. Wasatch Front Water Quality Council. Pp. 353. October 24.

Richards, D.C. 2018. Spawning Temperatures and Early Life State Expectations of Fish Species in Relation to Water Temperatures in the Lower Jordan River and State Canal. Wasatch Front Water Quality Council. Pp. 355. September 27.

Richards, D.C. 2018. Surplus Canal Native Mussel Survey 2018: As it Relates to the Central Valley Water Reclamation Facility Discharge. Wasatch Front Water Quality Council. Pp. 32. May 3.

Richards, D.C. 2019. Upper Jordan River Macroinvertebrate Assemblage. Wasatch Front Water Quality Council. Pp. 28. August 6.

Richards, D.C. 2020. An Ecological Evaluation of a Portion of the Lower Jordan River and Entire State Canal, Utah in Reference to Ammonia Criteria Standards for South Davis Sewer District: Focus on Fish Early Life Stages (ELS) in Relation to March Water Temperatures. South Davis Sewer District and Utah Division of Water Quality. Pp. 139. January 30.

Richards, D.C. 2020. Presumed Absence of Salmonid Reproductive Success in the Upper Jordan River in 2019-2020 based on Electrofishing Surveys and Probability of Detection Models. Pp. 6. March 1.

Richards, D.C. and T. Miller. 2019. Apparent Extinction of native mussels in Lower Mill Creek and Mid-Jordan River, Utah. Western North American Naturalist 79(1) pp. 72-84.

Richards, D.C. and T.G. Miller. 2018. Early Life Stage Fishery Survey in the Lower Jordan River and State Canal during the 'Shoulder' Month of October 2018. South Davis Sewer District and Utah Division of Water Quality. Pp. 38. October 29.

Richards, D.C., J. Holcomb, H. Finely, and F. Fluckiger. 2019. Lower Jordan River Mollusk Survey as it Relates to South Davis Sewer District South Plant Effluent. Progress Report 2019. Wasatch Front Water Quality Council. Pp. 10. January 2.

Richards, D.C., J. Holcomb, H. Finely, and F. Fluckiger. 2020. State Canal Mollusk Survey as it Relates to South Davis Sewer District North Plant Effluent. Progress Report 2020. Wasatch Front Water Quality Council. Pp. 8. January 3.

Richards, D.C., T. Miller, J. Holcomb, H. Finely, W.D. Robinson, and F. Fluckiger. 2020. Upper Jordan River, UT Fish Survey Winter 2019/2020. Progress Report. South Valley Water Reclamation Facility and Jordan Basin Water Reclamation Facility. Pp. 47. February 25.

Richards, D.C., T. Miller, J. Holcomb, H. Finley, W.D. Robinson, and F. Fluckinger. 2019. Ecological Assessment of Upstream Sections of the Jordan River, UT in Relation to Water Temperature Designated Beneficial Use Reevaluation. South Valley Water Reclamation Facility, Jordan Basin Water Reclamation Facility, and Utah Division of Water Quality. Pp. 103. October 16.

MILLER AND RICHARDS 2019

Follstad Shah, J., R. Smith, R. Gabor, S. Weintraub, Y. Jameel, and M. Navidomskis. Chapter 6: Nitrogen sources and transformations and Microbial community response to energy and nutrient availability in the Jordan River. Wasatch Front Water Quality Council. August.

Miller, T. 2019. Chapter 1: An Assessment of the Jordan River Using the Stream Visual Assessment Protocol. Wasatch Front Water Quality Council. Pp. 15. August.

Miller, T. 2019. Chapter 5 Sediment Oxygen Demand, Nutrient Flux, Organic matter processing and Methane Release from the Lower Jordan River. Wasatch Front Water Quality Council. Pp. 19. August.

Richards, D.C. 2019. Chapter 7: Jordan River Phytoplankton Assemblages: Rushforth and Rushforth 2009 Data Revisited. Pp. 47. August.

Richards, D.C. 2019. Chapter 8: Real and Perceived Macroinvertebrate Assemblage Variability in the Jordan River, Utah can Affect Water Quality Assessments. Wasatch Front Water Quality Council. Pp. 37. August.

Richards, D.C. 2019. Chapter 9: Is Reliance on a Single Bioassessment Metric for Assessing Water Quality in Utah's Rivers and Stream Prudent? Wasatch Front Water Quality Council. Pp. 39. August.

Richards, D.C. 2019. Chapter 10: Jordan River Macroinvertebrate Assemblages: Preliminary Findings. Wasatch Front Water Quality Council. Pp. 101. August.

Richards, D.C. 2019. Chapter 11: Upper Jordan River Macroinvertebrate Assemblage: November 2018. Wasatch Front Water Quality Council. Pp. 29. August.

This page is intentionally left blank.

APPENDIX B: RESPONSE TO COMMENTS

Comment #	Commenter Name	Comment	Response
1	Richards (comment letter, p. 1, par 3. and p. 3, par 3)	In my opinion, the most important outcome of Cirrus Ecological Solutions review and synthesis was the acknowledgement that two taxa, the Asian clam *Corbicula* sp. and the New Zealand mudsnail, *Potamopyrgus antipodarum*, likely govern many of the ecosystem functions in the now analog Jordan River, spatially and temporally. … 1. We have acknowledged that Corbicula and to a lesser extent NZMS are driving nutrients, DO, water quality, and the food web in the Jordan River and these rates vary spatially and temporally.	These comments overstate the results reported in the synthesis regarding the ecosystem influence of the Asian clam (Corbicula) and New Zealand mudsnail (NZMS) on the Jordan River. The synthesis does not conclude or imply that Corbicula or NZMS likely govern or drive primary ecosystem functions that influence DO in the lower Jordan River. The research synthesis reviewed recent Jordan River scientific research that contribute understanding of impaired DO conditions and support to the upcoming Phase 2 TMDL. Results from Richards (2018) were presented in sections 2.1.3 and 2.2.2. Some of these results provide rough density estimates of Corbicula and NZMS in the Jordan River between 5400 South and 900 South. Other results include literature values of filtering rates for surrogate macroinvertebrate species. Production of excreted OM, oxygen consumption, and CO_2 respiration are also mentioned as a negative impact from Corbicula and NZMS. The last paragraph in section 2.1.3 states "Based on literature values of filtering rates and estimates of density, *Corbicula* could potentially filter large volumes of the Jordan River depending on the estimated density of this species (Richards 2018)." Again, this statement does not imply that Corbicula or

Comment #	Commenter Name	Comment	Response
			NZMS govern or drive primary ecosystem functions in the Jordan River. Considerable uncertainty currently exists regarding the actual density, distribution, and net impact (i.e. positive or negative) of Corbicula and the NZMS. The Phase 2 TMDL will consider all available, scientifically defensible information related to DO impairment in the lower Jordan River.
2	Richards (comment letter, p. 2, par's 2, 4, and 5)	I highly recommend __not__ using values in 'A snail, a clam, and the River Jordan' report. Those values should be considered 'ballpark' or 'jumping off' values to elicit actual *Corbicula* and *Potamopyrgus* bioenergetics, nutrient cycling, and food web dynamics studies in the Jordan River. … I did find discrepancies in Newall et al (2005) mussel consumption and excretion report. Consumption and excretion values in Newall were composited from different studies. Some data were based on oysters in Chesapeake Bay, not directly relevant to Jordan River. Upon further review, consumption rates are likely 1 to 2 orders of magnitude higher than what I reported, and a more recent find was that excretion rates could be 3 times less than what I reported. However, these values are from other ecosystems and should not be used in any Jordan River models. … I am updating the 'A snail, a clam, and the River Jordan' report with these revised values and emphasizing that those values are from other locations and sometimes other species. This updated version will be part of a mollusk compendium for the Jordan River-Utah Lake drainage that OreoHelix	Comment noted. Please provide an updated copy of Richards (2018) when it is available.

Comment #	Commenter Name	Comment	Response
		Ecological and WFWQC anticipate having available by end of August 2020.	
3	Richards (comment letter, p. 2, par 3)	Recommendations 1. Conduct field research to verify and update estimated values calculated for Jordan River from literature and presented in this report. Specifically, estimate densities of *Corbicula* and *Potamopyrgus* in sections not surveyed by OreoHelix Consulting and The Wasatch Front Water Quality Council. Conduct in situ experiments using the most up to date methods for estimating: carbon, nitrogen, and phosphorous filtering and consumption rates, excretion rates, and carbon fixation rates, etc. (see Same and Olenin 2005). Relate these findings to water chemistry values in the Jordan River and determine the effects of the clam and snail. <u>Update all Jordan River reports with this new information. Inform researchers and managers."</u>	This recommendation was summarized and added to section 5.5 (see Oxygen Demand, bullet 6).
4	Richards (comment letter, p. 3, par 2)	I cannot emphasize enough that the Jordan River was and continues to be a molluskan hot spot in the Western USA and a Utah natural heritage; and that these native and invasive taxa are major ecosystem engineers and regulators of water quality. I do not agree with Cirrus that mollusk effects (or other macroinvertebrates) have been fully or even partially addressed in the DWQ models based on Hogsett SOD values. A more useful and ecologically relevant model would seek to understand these effects in more detail to better understand the contribution of benthic invertebrates to SOD and for DWQ to properly manage the Jordan River.	The Phase 2 TMDL is required to quantify sources of OM as a pollutant of concern that leads to DO impairment in the lower Jordan River. Research has identified SOD as a significant loss of DO and therefore, must be addressed in the lower Jordan River. Field methods for measuring SOD would include the influence of benthic invertebrates on DO concentration. This approach meets TMDL requirements. If benthic invertebrates are determined to contribute to a significant loss of DO, the SOD measurements used in the TMDL could be partitioned accordingly.

Comment #	Commenter Name	Comment	Response
5	Richards (comment letter, p. 3, par 3)	2. These two taxa are surrogates for extinct species that in the past cleaned the Jordan River. Without these two taxa (and the remaining natives), the JR would be putrid at certain times of year.	Thank you for your comment.
6	Richards (comment letter, p. 3, par 3)	3. The urgency of conducting experiments and measurements on the ecological and water quality contributions of these two species in the Jordan River ecosystem.	Comment noted. See recommendations in section 5.5 Oxygen Demand for measuring densities of Corbicula and Potamopyrgus in the Jordan River and estimating their water quality influence.
7	Richards (comment letter, p. 3, par 4)	It would be helpful if Cirrus/DWQ included a sentence or paragraph on why low DO is a problem. We all realize that low DO can be harmful to fisheries and other biota, so a justification statement would benefit.	See edits to section 1, paragraph 2.
8	Richards (comment letter, p. 3, par 5, and p. 4, par 1)	Macroinvertebrates most certainly do influence Jordan River water quality, however, macroinvertebrates are more than *Corbicula* and *Potamopyrgus*. There are dozens of benthic invertebrate taxa in the river, each with a unique ecological niche and water quality requirements. Subsequently, each taxon provides unique insights into water quality conditions. More importantly, however, macroinvertebrates **are** water quality as the Clean Water Act and DWQ designated beneficial uses clearly state… .	We agree that one objective of the Clean Water Act is to protect aquatic life (including shellfish and other benthic invertebrates). As explained in section 1, the Class 3B and 3D beneficial use assigned to the LJR is designed to protect warm water fish species and other wildlife that depend on an aquatic environment.
9	Richards (comment letter, p. 4 par 2)	Benthic macroinvertebrates obviously contribute to oxygen demand in the water column and sediments, however benthic invertebrates **are** the reason DWQ was relegated with developing TMDLs (see next section).	Again, we agree that one objective of the Clean Water Act is to project shellfish and benthic invertebrates. However, Utah DWQ has responsibility for developing TMDLs to protect all beneficial use categories in Utah. Regarding the Jordan River, this includes

Comment #	Commenter Name	Comment	Response
			many aspects of water quality in addition to benthic invertebrates.
10	Richards (comment letter, p. 4 par 2)	Benthic macroinvertebrate contributions, particularly positive contributions, need to be included in any TMDL model. One example of benthic invertebrate effects on SOD are chironomid larvae. …DWQ considers chironomids to be indicators of water quality impairment without taking into account their positive contribution to increased DO in the sediments. It is imperative for us to develop a better understanding of the ecologies of these midge taxa in the river to better manage it. At this time, DWQ DO TMDL models need to address benthic invertebrate influences on DO, or at least acknowledge model uncertainty due to this lack of understanding. DWQ models are heavy on chemistry data inputs, but sorely lacking actual biological and ecological inputs. Without such, models are only half useful.	The Phase 2 TMDL must address all pollutant sources that contribute to DO impairment due to low concentrations of DO. To the extent that chironomid larvae or other benthic invertebrates increase DO, these benefits can be discussed in the TMDL. Based on results in Richards (2018), *Corbicula* and *Potamopyrgus* also have negative impacts on water quality due to respiration and excretion. The net benefit of these species in the LJR is currently unknown. If additional research shows a positive net benefit, intentionally increasing densities of these invasive species would not be used to improve DO in the LJR.
11	Richards (comment letter, p. 4 par 3)	Cirrus stated that "Despite sufficient light and nutrients, measurements do not show additional growth of benthic algae in the LJR". In addition to unstable and unsuitable substrate, much of this lack of an additional benthic algal growth observation could be due to grazing by benthic invertebrates and fish benthic algaevores. Snails and other benthic invertebrates in the Jordan River are voracious algaevores, as are the highly abundant native Utah Sucker. …An incomplete explanation for the unobservable changes to benthic algal growth by our group could be the consequence of incomplete ecological knowledge and understanding of this grazing induced feedback loop.	See edits to section 2.1.1, paragraph 6.

Comment #	Commenter Name	Comment	Response
12	Richards (comment letter p. 5 and p. 6 par's 1-2)	5 Ecological integrity, the Clean Water Act, and UDWQ designated beneficial use.	We appreciate your response regarding the biological integrity of the Jordan River; however, this comment is out of scope for the purpose of the research synthesis. We will consider this comment as we move forward with the Jordan River DO TMDL as well as other options for riverine management including use attainability analysis and/or site-specific criteria.
13	Richards (comment letter p. 6 par's 3-5 and p. 7 par's 1-3)	6 Jordan River: an analog system.	We agree that the Jordan River has undergone drastic changes resulting in a system much different than its former self. The current condition of the river will be considered when determining management options.
14	Richards (comment letter, p. 7 par 5)	Obviously the easiest and most prudent remedy for DO impairment in the Jordan River is to flush OM downstream out of the river system, somewhat simulating a natural hydraulic regime. …Dredging OM from the Lower Jordan River would be a second option and also easily accomplished. Because the river is a poor analog of its former self, these two options would not negatively affect the now analog Jordan River's ecological integrity, which has all but been lost, or even its designated beneficial use.	Recent research has examined opportunities for managing flow with the intent to improve DO in the LJR. Some results of this research have recommended periods of additional flow during spring to mimic natural hydrology and the flushing actions that naturally accompany these events. These actions could have corresponding impacts on property owners and downstream water users and must be addressed prior to any field assessment or change in flow management.
15	Richards	It is my opinion that the only reason for TMDL development for any type of pollutant, including DO, is to protect designated beneficial uses, e.g. "Class 3B -- Protected for	Designated beneficial use and water quality standards assigned to the LJR are designed to protect both existing and potential ecological

Comment #	Commenter Name	Comment	Response
	(comment letter, p. 7 par 6 and p. 8 par 1)	warm water species of game fish and other warm water aquatic life, including the necessary aquatic organisms in their food chain" in the Jordan River. Without a more comprehensive ecological understanding of these biota, then we may not be able to fully justify TMDL values.	health for these river segments. The Phase 2 TMDL will allocate loads that will protect these uses and meet state and federal regulations.
16	Follstad Shah (comment letter, p. 1 par 2)	Fig. 2-1: It would be easier to interpret patterns if cyanobacteria were plotted on an axis separate from other algal species. Also, axes scales should be similar for each month. Most of the variation in algal species biovolume in July and August is masked by the scale of the y-axes.	Agree. Text in section 2.1.1, paragraph 4 notes this. A request for original data has been made for the original data in section 5.5 Data Gaps – Total OM and OM pollutant sources. A more detailed assessment will occur if a copy of original data records is provided.
17	Follstad Shah (comment letter, p. 1 par 3)	Legend of Fig. 2-2: Instream processing could be better defined here. Evidence of instream processing occurs when measured river nutrient load is less than the cumulative nutrient load from effluent inputs.	See edits to Figure 2-2 caption.
18	Follstad Shah (comment letter, p. 1 par 4)	p. 10 in general: much of this text needed editing to be accurate with respect to theoretical underpinnings and empirical evidence of ecoenzyme activities in the environment. In addition, some of the conclusions included in the report were taken from older presentations that do not reflect the evolution of the authors' interpretations. The biggest issue is that the concept of N-limitation is not valid in this N rich system, both with respect to organic and inorganic forms of N (**Mori 2020**). Rather, microbial communities seem to be responding to the most abundant form of organic matter in the system, which appears to be proteins that may be derived from algae or effluent. Coupling between C and N activities and consistent activities of NAG, which targets chitin and peptidoglycan to acquire both C and N, also	See edits to section 2.1.2, paragraphs 4–9.

Comment #	Commenter Name	Comment	Response
		suggest microbes are utilizing algal-derived organic matter during baseflow conditions. I have edited the text in each paragraph with this in mind. Specific rationale is provided in the comments below.	
19	Follstad Shah (comment letter, p. 1 par 5)	p. 10, first and second sentences: The process of decomposition does not release enzymes to the environment. Rather, bacteria secrete enzymes into the environment to degrade complex organic compounds. The first sentence should be edited for better clarity. The second sentence also is unclear. It should explain that the balance of enzymes secreted by bacteria to acquire energy or nutrient resources depends on the availability of C, N, and P in the environment relative to the stoichiometric balance of C, N, and P within microbial biomass.	See edits to section 2.1.2, paragraph 3.
20	Follstad Shah (comment letter, p. 1 par 6 and p.2 par 1)	p. 10 second paragraph, first sentence: It is true the quality of organic matter can be inferred from EEA analysis in two ways: 1) the ratio of β-1,4-glucosidase activity (BG; expressed to hydrolyze more labile C – e.g., cellulose) to phenol (or peroxidase) activity (POX; expressed to oxidize more recalcitrant C – e.g., lignin) (Sinsabaugh and Follstad Shah 2012), or 2) measurement of dehydrogenase activity, which is a proxy for microbial community respiration rate (**Hill et al. 2012**). The latter has not been measured in the Jordan River. Oxygen demand can be high for both labile and more recalcitrant forms of organic matter. It depends on the composition of the extant microbial community and its ability to produce the necessary enzymes in quantities required to degrade the dominant form of organic matter available. I added text to this paragraph to describe which enzymes are	See edits to section 2.1.2, paragraph 4.

Comment #	Commenter Name	Comment	Response
		used to acquire energy or nutrients from different sources of organic matter.	
21	Follstad Shah (comment letter, p. 2 par 2)	p. 10 third paragraph: Enzyme expression is sometimes tricky to interpret. Sometimes elevated activity indicates limitation, sometimes it indicates resources switching to an abundant resource. I added to text to make this point.	See edits to section 2.1.2, paragraph 5.
22	Follstad Shah (comment letter, p. 2 par 3)	p. 10 fifth paragraph: I edited this paragraph for greater accuracy related to the interpretation of 'N-limitation' and relationships between C and N acquisition by microbial communities.	See edits to section 2.1.2, paragraph 7.
23	Follstad Shah (comment letter, p. 2 par 4)	Fig. 2-6: This figure should indicate that the graphs were first published in **Sinsabaugh et al. 2009**.	See edits to Figure 2-6 caption.
24	Follstad Shah (comment letter, p. 2 par 5)	Fig. 2-7: Some of the axis titles are truncated. I can provide a better version if desired.	See update Figure 2-7, copied from Follstad Shat et al. (2019b).
25	Follstad Shah (comment letter, p. 2 par 6)	p. 15 first paragraph: The report correctly states that EEM data (i.e., fluorescence index, or FI) indicates DOM is microbially produced, and rather labile (Follstad Shah et al. 2017). We also show a similar trend in EEA data. Specifically, BG activity rates are 2 orders of magnitude greater than POX activity rates (Follstad Shah et al. 2019b – as cited in report). This indicates that the microbial community is utilizing more labile forms of polysaccharides relative to more recalcitrant forms of carbon, such as lignin and tannins. This point was not included anywhere in the	Additional text added to concluding paragraph of section 2.1.2 that references section 3.3 and partially introduces the labile nature of Jordan River DOM. Also see edits to section 3.3, paragraph 5 where text was added to emphasize microbial use of labile polysaccharides relative to recalcitrant forms of C (e.g. lignin and tannins).

Comment #	Commenter Name	Comment	Response
		report. (However, all of this is already described on p. 34, so may not be needed here.)	
26	Follstad Shah (comment letter, p. 2 par 7)	p. 32 last paragraph: The discussion of specific FI values in the Jordan River relative to other locations should be interpreted with caution. We discovered, through these unusually high values, that different laboratory equipment used to calculate EEMs can produce different values. Rachel Gabor, the PI investigating DOM quality using EEMs, has been working to quantify the differences between machine outputs, using a consistent set of samples for analyses. Nonetheless, relative change in FI along the Jordan River flowpath can be compared amongst study sites within the Jordan River system (e.g., last paragraph of p. 33).	See edits to section 3.3, paragraph 3.
27	Follstad Shah (comment letter, p. 2 par 8)	p. 33 last paragraph: water isotopes indicated that effluent is the dominant source of water to the LJR in fall. This paragraph notes that FI values are elevated in fall relative to other seasons, dismissing detritus as the major source of DOM to the river. However, higher values of FI in fall are consistent with effluent as the dominant source of water to the river, as DOM in effluent has been highly processed by microbes during the wastewater treatment.	See edits to section 3.3, paragraph 4.
28	Follstad Shah (comment letter, p. 2 par 9)	p. 50 first bullet under Aerobic Decomposition: Again, I caution interpreting EEA data in terms of limitation, given that high LAP activity rates more likely reflect microbial utilization of an abundant substrate. In short, organic N is NOT limiting. Variation in P acquisition is more variable, and likely a response to available forms of organic C and N.	See edits to section 5.1, bullet 1 under Aerobic Decomposition in the Water Column.

Comment #	Commenter Name	Comment	Response
29	Follstad Shah (comment letter, p. 3 par 1)	p. 51 last bullet under Sediment Oxygen Demand: EEA ratios in the sediment behave more similarly to the traditional paradigm, than patterns observed in the water column. Thus, the last phrase in this sentence is misplaced.	See edits to section 5.1, bullet 3 under Sediment Oxygen Demand and Nutrient Processes.
30	Follstad Shah (comment letter, p. 3 par 2)	p. 53 first bullet: Our efforts could not distinguish between 2H signatures in riparian vegetation and biofilms, due to methodological issues that were avoided by Kelso et al. (2018). Hence, this sentence is not accurate. Kelso et al. (2018) were able to make this distinction.	Agree. This bullet has been deleted (section 5.2, bullet 9).
31	Follstad Shah (comment letter, p. 3 par's 4–6)	**Relevant references not currently in report:**	Several of these references were used as part of text edits.
32	Utah Department of Transportation (email, item 1)	Will the impacts of OM from Utah Lake be quantified similarly to how the stormwater OM impacts were considered? The relative percent contribution of Utah Lake to the different OM particle sizes was given, but the links were less clearly quantified for what impacts the OM load from the lake has on BOD and SOD downstream.	The Phase 1 TMDL identified Utah Lake as one of several significant sources of Total OM contributing to DO impairment in the lower Jordan River. Research results included in the synthesis (e.g. Kelso 2018 and Follstad Shah et al. 2019a) have further defined the potential of the Lake to contribute OM loads and low DO. We anticipate the WASP model, along with supporting research results, will be used to define a linkage between OM sources (including Utah Lake) and oxygen demand (i.e. BOD and SOD) in impaired segments of the lower Jordan River.

Comment #	Commenter Name	Comment	Response
33	Utah Department of Transportation (email, item 2)	In table 5-1, OM Sources, Implications for Phase 2 TMDL it states: "Stormwater OM should be accounted for during dry and wet (storm event) weather". We were looking for clarification as to what was intended with stormwater inputs during dry weather? Is this referring to outfalls that have unidentified illegal connections or groundwater intrusion so that there is some level of baseflow from the outfall? Or is there some other way that it is thought stormwater could be contributing during dry weather?	We are aware that some stormwater outfalls to the Jordan River contribute flow during dry weather (i.e. non-storm periods). The source of this flow is currently unknown and should be accounted for separately from the flow and OM loading generated during storm events. As noted, the source of dry weather flow could be unidentified illegal connections to stormwater drains or groundwater inflow.
34	Juhn-Yuan Su (email, first bullet, second paragraph)	Parameterization: The hydraulic parameterization in the Jordan River WASP appears dependent upon the routing method applied that further affects calculations for flow, residence times, and other hydraulic parameters pertinent for the WASP model. The Jordan River WASP currently employs the **kinematic wave routing method**, enabling the user to specify the typical parameters for applying the Manning's flow (e.g., Manning's roughness coefficient, bottom width, channel slope, etc.), along with the minimum depth and depth-velocity exponents.	Comment noted. The flow routing method selected appears to be a reasonable and appropriate choice for modeling flow in the Jordan River watershed. Given that the WASP model is dynamic and includes periods of dry weather and storm events, our concern is that modeled residence times and flow velocities are representative during critical times when significant OM loading occurs. See text edits to section 4.3.1, paragraph 2.
35	Juhn-Yuan Su (email, first bullet, third paragraph)	Selection of the Kinematic Wave Method: The **kinematic wave routing method** is selected for allowing the user to directly specify the flow into a single segment (e.g., point source, nonpoint source, etc.), with the first flow function specifying flow segmentation for the entire model. If other routing methods are selected (e.g., flow routing, stream routing, dynamic wave, diffusive wave), then the user will need to **define every additional flow from the point/nonpoint source segment to out of the Jordan River**	Comment noted. See previous response and text edits to section 4.3.1, paragraph 2.

Comment #	Commenter Name	Comment	Response
		system (e.g., to "Boundary"). … If other routing methods are selected (e.g., other than the kinematic wave method), then one failing to trace the additional flow for ensuring that such additional flow has left the system will instigate the model segment to continuously increase in flow volume, further causing *all* water quality concentrations to decrease and approach to 0.	
36	Juhn-Yuan Su (email, first bullet, fourth paragraph)	Limitations of Flow Applications in WASP: One major limitation of WASP involves the implementation of a "boxed" model, yielding a rectangular channel (e.g., bottom width, depth) as compared to a trapezoidal channel allowed in the Qual2K models. Such limitations may need to be discussed with members associated with the development of WASP (e.g., EPA) for potentially enhancing the hydraulic parameterizations under such routing methods.	Comment noted. Future model validation will consider channel geometry and any potential influence on flow routing and selection of hydraulic parameters.
37	Juhn-Yuan Su (email, second bullet)	Model Calibration during Summer Periods: The model calibration is suggested to focus primarily during the summer periods for which low flows and elevated temperatures are typically observed over the Jordan River as the prediction of flow volumes by the Jordan River WASP significantly affects the water quality predictions yielded for other constituents (e.g., organic matter, nutrients, etc.). (Note: The model calibration conducted upon the Jordan River WASP is applied for the purposes of the EPA Project Work at the University of Utah, which involves assessing the Jordan River watershed performance under climatic effects and population growth characteristics *per season*. … Furthermore, the frequency of measured water quality data from the UDWQ AWQMS database employed for the Jordan River WASP model calibration work during	Comment noted. We recognize the focus of model calibration required by the EPA and University of Utah is different than what is needed to meet TMDL requirements. See edits to section 4.3.5, paragraph 1.

Comment #	Commenter Name	Comment	Response
		the summer times appears to have such focus (e.g., model calibration on the summer times) seem rather unrealistic (e.g., low-frequency measured water quality data during the summer months for several water quality constituents).)	