



Effects of Flow Diversion on Snake Creek and its Riparian Cottonwood Forest, Great Basin National Park

Natural Resource Report NPS/GRBA/NRR—2020/2104



**ON THIS PAGE**

Upstream view of the Snake Creek watershed. The riparian cottonwood forest downstream from the dewatered reach is visible.
Credit: D.M. Schook

ON THE COVER

Study of the Snake Creek river corridor in Great Basin National Park.
Credit: D.M. Schook

Effects of Flow Diversion on Snake Creek and its Riparian Cottonwood Forest, Great Basin National Park

Natural Resource Report NPS/GRBA/NRR—2020/2104

Derek M. Schook^{1,2}, David J. Cooper¹, Jonathan M. Friedman³, Steven E. Rice², Jamie D. Hoover^{2, 4}, Richard D. Thaxton^{1,3}

¹ Department of Forest and Rangeland Stewardship
Colorado State University
Fort Collins, CO

² Water Resources Division
National Park Service
Fort Collins, CO

³ US Geological Survey
Fort Collins Science Center
Fort Collins, CO

⁴ Front Range Community College
Longmont, CO

April 2020

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado, publishes a range of reports that address natural resource topics. These reports are of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

The Natural Resource Report Series is used to disseminate comprehensive information and analysis about natural resources and related topics concerning lands managed by the National Park Service. The series supports the advancement of science, informed decision-making, and the achievement of the National Park Service mission. The series also provides a forum for presenting more lengthy results that may not be accepted by publications with page limitations.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner.

This report received formal, high-level peer review based on the importance of its content, or its potentially controversial or precedent-setting nature. Peer review was conducted by highly qualified individuals with subject area technical expertise and was overseen by a peer review manager.

Views, statements, findings, conclusions, recommendations, and data in this report do not necessarily reflect views and policies of the National Park Service, U.S. Department of the Interior. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the U.S. Government.

This report is available in digital format from the [Natural Resource Publications Management website](#). If you have difficulty accessing information in this publication, particularly if using assistive technology, please email irma@nps.gov.

Please cite this publication as:

Schook, D. M., D. J. Cooper, J. M. Friedman, S. E. Rice, J. D. Hoover, and R. D. Thaxton. 2020. Effects of flow diversion on Snake Creek and its riparian cottonwood forest, Great Basin National Park. Natural Resource Report NPS/GRBA/NRR—2020/2104. National Park Service, Fort Collins, Colorado.

Contents

	Page
Figures.....	viii
Tables.....	xii
Abstract.....	xiv
Acknowledgements.....	16
1. Introduction.....	17
1.1 Background.....	17
1.2 Study site: The Snake Creek drainage basin and water diversion pipeline	18
1.3 Study approach.....	19
2. Study Site: Geology, Weather, and River Flow.....	22
2.1 Study Reaches	22
2.1.1 Dewatered (DW) Reach	22
2.1.2 Reference Reaches.....	24
2.2 Geology and Hydrogeology	25
2.3 Weather and Climate	27
2.4 Streamflow	28
3. Cottonwood Forest Condition: Demography, Canopy Vigor, and Tree Mortality	32
3.1 Introduction	32
3.2 Cottonwood Demography	32
3.2.1 Survey of Live and Dead Trees	32
3.2.2 Cottonwood Size and Age Distributions	33
3.3 Cottonwood Canopy Vigor.....	38
3.3.1 Methods	38
3.3.2 Results and Discussion	39
3.4 Cottonwood Dieback and Mortality	42
3.4.1 Patterns of tree mortality across reaches	42
3.4.2 Within-tree branch dieback	48

4. Tree Ring Chronologies	52
4.1 Introduction	52
4.2 Methods	52
4.2.1 Sample collection and processing	52
4.2.2 Tree ring chronologies.....	53
4.3 Results	54
4.3.1 Descriptive statistics.....	54
4.3.2 The dewatered (DW) reach chronology	55
4.3.3 DW and reference reach chronologies.....	56
4.3.4 Tree ring growth by canopy class.....	59
4.3.5 Tree ring growth by age group	61
4.3.6 Tree ring growth in DW subreaches.....	62
4.4 Synthesis of Tree Ring Chronologies.....	64
5. Stable Isotopes in Tree Rings	65
5.1 Introduction	65
5.2 Methods	65
5.2.1 Sample preparation.....	65
5.2.2 Data analysis.....	66
5.3 Results	67
5.3.1 $\delta^{13}\text{C}$ through time	67
5.3.2 Climate connections to $\delta^{13}\text{C}$	69
5.3.3 Relationships between tree ring width and $\delta^{13}\text{C}$	71
5.4 Discussion.....	72
6. Hydroclimatic Drivers of Tree Ring Width	74
6.1 Introduction	74
6.2 Methods	74
6.3 Results and Discussion	75
6.3.1 Tree rings and flow, precipitation, and temperature.....	75

6.3.2 Comparing pre- and post-pipeline periods	75
6.3.3 Individual trees	76
7. Cottonwood Ecophysiology	79
7.1 Introduction	79
7.2 Methods	79
7.2.1 Field-based leaf water potential.....	79
7.2.2 Percent loss of leaf hydraulic conductance.....	80
7.3 Results	81
7.3.1 Field-based leaf water potential.....	81
7.3.2 Leaf hydraulic conductivity.....	83
7.4 Discussion.....	84
8. Aerial Photos.....	86
8.1 Introduction	86
8.2 Methods	86
8.2.1 General approach.....	86
8.2.2 Riparian tree cover through time	88
8.2.3 NDVI	88
8.3 Results and Discussion	90
8.3.1 Riparian tree cover through time	90
8.3.2 NDVI through time.....	93
9. Stand-Level Cottonwood Production.....	97
9.1 Introduction	97
9.2 Methods	97
9.3 Results	98
9.3.1 Aerial Photo approach	98
9.3.2 Tree-count approach.....	99
9.4 Discussion.....	100
10. Groundwater Monitoring	102

10.1 Introduction	102
10.2 Methods	102
10.3 Results	105
10.3.1 Water levels	105
10.3.2 Well borehole extractions	113
10.4 Discussion.....	114
11. Near-Surface Geophysics.....	116
11.1 Introduction	116
11.2 Methods	116
11.3 Results and Discussion	118
11.3.1 Transect line 5	118
11.3.2 Transect line 6	118
11.3.3 Transect line 7	120
11.3.4 Transect lines 1 and 2	121
11.3.5 Implications and correspondence to groundwater wells.....	123
12. Channel and valley morphology	124
12.1 Introduction	124
12.2 Methods	124
12.2.1 Study reaches.....	124
12.2.2 Channel cross-sections	126
12.2.3 Valley setting.....	126
12.3 Results	127
12.3.1 Channel cross-sections	127
12.3.2 Valley setting.....	129
12.4 Discussion.....	131
13. Instream wood.....	133
13.1 Introduction	133
13.2 Methods	133

13.3 Results	134
13.4 Discussion.....	136
14. Summary and Synthesis.....	138
14.1 Summary of each research investigation.....	138
14.2 Synthesis.....	141
15. References.....	144
16. Appendix.....	158
16.1 Tree ring chronologies.....	158
16.2 Aerial photos	159

Figures

	Page
Figure 1.1. Snake Creek and the study reaches in Great Basin National Park.....	19
Figure 2.1. Upstream views of Snake Creek in the dewatered (DW) reach.	22
Figure 2.2. Pictures of cottonwood stands found along Snake Creek.....	24
Figure 2.3. Southern Snake Range geologic cross-section.	26
Figure 2.4. Climograph of 1981–2010 monthly average precipitation, daily maximum temperature, and daily minimum temperature.	27
Figure 2.5. Mean temperature, total precipitation, and vapor pressure deficit from PRISM displayed as 5-year running averages.	28
Figure 2.6. Snake Creek and Cleve Creek hydrographs for the period of overlap, water years 2011–2017.	30
Figure 2.7. Boxplot of the distribution of mean monthly flows at Cleve Creek, water years 1960–2017.	30
Figure 2.8. Mean monthly flow for the infilled Cleve Creek dataset for water years 1960–2017.....	31
Figure 3.1. Tree core collection pictures.....	34
Figure 3.2. Number of trees by tree diameter for each size class in the DW reach.	36
Figure 3.3. Age-Diameter relationship for all live trees sampled in the DW reach.....	36
Figure 3.4. Proportion of cored trees establishing by year.	37
Figure 3.5. Example pictures of cottonwoods in each canopy class.	39
Figure 3.6. Percent live tree canopy for all standing cottonwoods across reaches.	40
Figure 3.7. Percent live canopy among living trees in three age classes from the DW reach.....	41
Figure 3.8. Leaf area index for the six canopy classes.....	42
Figure 3.9. First and last year of growth for sampled standing and fallen dead trees in the DW reach.	45
Figure 3.10. Death dates for standing dead trees.	46
Figure 3.11. Death dates from all standing and fallen trees in the Dewatered reach.	47
Figure 3.12. Tree ring growth as a function of years before tree death.	47

Figure 3.13. Photos of two example dissected trees showing the location of cross-dated branches and their death dates.	51
Figure 3.14. Death dates of the 158 sampled branches from 13 trees.	51
Figure 4.1. Basal area increment as a function of tree age.	54
Figure 4.2. The five cottonwood subreaches in the dewatered reach.	54
Figure 4.3. Tree ring chronology for the Dewatered reach, 1890–2016.	56
Figure 4.4. Tree ring chronologies from the DW and four reference reaches from 1901 to 2016.	58
Figure 4.5. Tree ring chronologies for the six canopy classes in the DW reach.	60
Figure 4.6. Dewatered reach chronologies constructed from living trees standing dead trees	60
Figure 4.7. Decadal mean BAI for DW trees and the standard deviation among the six canopy classes.	61
Figure 4.8. BAI through time for three age classes of Dewatered reach cottonwoods.	62
Figure 4.9. Tree ring chronologies for the five DW subreaches.	63
Figure 5.1. Annual tree ring $\delta^{13}\text{C}$ values from DW (live and dead), RU, and RD study reaches for 1942–2016.	68
Figure 5.2. Annual tree ring $\delta^{13}\text{C}$ relative to baseline for DW live, DW dead, RU, and RD trees.	69
Figure 5.3. Trees plotted on the first two principal components of a k-means cluster analysis based on $\delta^{13}\text{C}$ of tree rings.	69
Figure 5.4. Correlations for tree ring width or $\delta^{13}\text{C}$ with flow and precipitation in the DW reach (1962–2016).	71
Figure 6.1. Tree ring correlations to flow, precipitation, and temperature.	77
Figure 6.2. Annual tree ring width correlation to precipitation and temperature before and after the pipeline was installed.	78
Figure 6.3. Percent of DW reach trees with maximum correlations between ring width and flow, precipitation and temperature.	78
Figure 7.1. Leaf water potentials for cottonwoods in dewatered (DW) and reference downstream (RD) reaches for predawn and midday periods.	82
Figure 7.2. Leaf water potentials for cottonwoods in the DW and RD reaches between 06:00 and 13:00.	82

Figure 7.3. Percent loss of hydraulic conductance curves for leaves sampled from the DW and RD reaches.....	84
Figure 8.1. The Snake Creek aerial photo study reaches in 1946 and 2013.	87
Figure 8.2. Riparian area per photo year as a proportion of baseline area.....	91
Figure 8.3. Riparian area delineations from object-based image analysis.....	93
Figure 8.4. Average tree-level NDVI.....	95
Figure 8.5. NDVI values per canopy class.....	95
Figure 8.6. Effect sizes for NDVI means from vegetation groups in the focal area analysis.....	96
Figure 9.1. Stand basal area increment per river kilometer for the DW and Reference reaches using the air photo approach.	99
Figure 9.2. Stand-level basal area increment for cottonwoods by reach.....	100
Figure 10.1. Groundwater well Transect 3 as an example.	104
Figure 10.2. Locations of well and staff gauge transects along Snake Creek.....	104
Figure 10.3. Cross-section at Transect 1 and time-series of water table depths at Well 1 and Staff gauge 1 from April to September 2019.	108
Figure 10.4. Cross-section at Transect 2 and time-series of water table depths at Well 2, Well 3, and Staff gauge 2 from June to September 2019.	109
Figure 10.5. Cross-section at Transect 3 and time-series of water table depths at Well 4, Well 5, and Staff gauge 3 from September 2018 to September 2019.....	110
Figure 10.6. Cross-section at Transect 4 and time-series of water table depths at Well 6, Well 7, and Staff gauge 4 from April to September 2019.	111
Figure 10.7. Cross-section at Transect 5 and time-series of water table depths at Well 4, Well 5, and Staff gauge 3 from February to September 2019.	112
Figure 10.8. Surface water levels at the monitoring transects from April to September 2019.....	113
Figure 11.1. Geologic cross-section showing the approximate location of electrical resistivity survey lines and other features in the Snake Creek study area.	117
Figure 11.2. Upstream view of electrical resistivity profile at Transect 5.	118
Figure 11.3. Electrical resistivity inversion profile at Transect 6.	119
Figure 11.4. Researcher standing in a sinkhole below the Snake Creek pipeline inlet structure.....	120

Figure 11.5. Upstream view of the electrical resistivity inversion profile at Transect 7.	121
Figure 11.6. Electrical resistivity inversion profiles for longitudinal Transect 1 and valley cross-section Transect 2.	122
Figure 11.7. Distribution of cottonwoods upstream and downstream of lateral groundwater inputs at Squirrel Springs.....	122
Figure 12.1. The Snake Creek study reaches (RU, DW, RD) are separated by the pipeline inlet and outlet.....	125
Figure 12.2. Example images of Snake Creek and its valley bottom.....	125
Figure 12.3. An example portion of Snake Creek’s DW reach showing 20-meter valley bottom increments.....	127
Figure 12.4. Average width, depth, and bed material grain size for the RD, RU, and DW reaches.....	128
Figure 12.5. Boxplots showing differences in channel dimensions and grain size for the dewatered (DW), reference upstream (RU) and reference downstream (RD) reaches.....	128
Figure 12.6. Boxplots of channel dimensions and grain size with the study reaches where the DW reach is divided into six subreaches.	129
Figure 12.7. Relationships between Snake Creek channel dimensions and valley slope.	130
Figure 13.1. Example images of instream wood jams.	134
Figure 13.2. Instream wood jams in the three Snake Creek study reaches.	135
Figure 13.3. Volume of wood per jam in the study reaches.	136
Figure 16.1. Appendix: Alternative tree ring chronologies.	158

Tables

Page

Table 2.1. Description of the study reaches.	23
Table 2.2. Selected geologic units mapped within the study area.....	26
Table 3.1. Live or dead status of standing trees in the four primary study reaches.	33
Table 3.2. Live dewatered reach trees in each diameter size class.....	35
Table 3.3. Timing of whole tree and branch death for the dissected trees.....	50
Table 4.1. Dendrochronology descriptive statistics for tree ring chronologies by reach.....	55
Table 4.2. Average BAI for the study reaches for 1933–1961 and 1962–2016.....	57
Table 4.3. Mean BAI from the five DW subreaches for select periods.	63
Table 5.1. Description of the $\delta^{13}\text{C}$ isotope samples.	68
Table 5.2. Correlations between $\delta^{13}\text{C}$ and basal area increment (BAI) for the pre-pipeline (1942–1961) and post-pipeline (1962–2016) study periods.	72
Table 7.1. Summary of ecophysiology measurements made in the dewatered (DW) and reference downstream (RD) reaches.....	81
Table 7.2. Percent loss of conductivity for the DW and RD reaches.....	83
Table 8.1. Aerial photo metadata.	87
Table 8.2. Descriptions of the ten focal areas from NDVI analysis.....	90
Table 8.3. Riparian area per photo year in the Dewatered and Reference reaches.	92
Table 8.4. Linear models of mean NDVI value and percent live canopy per reach.	94
Table 10.1. Groundwater wells and surface water staff gauges with the number of days with measured water.	107
Table 10.2. Summary of the extracted material from well boreholes.	114
Table 11.1. Summary of Snake Creek electrical resistivity survey lines.	117
Table 12.1. Linear models of channel dimensions as a function of valley width.	131
Table 12.2. Proportion of the difference in channel dimensions between DW and reference reaches unexplained by valley confinement.	131

Table 13.1. Wood jam characteristics for the DW, RU, and RD reaches.	135
Table 16.1. Appendix: The multiscale segmentation algorithm used at two settings to identify coarse- and fine-scale objects in the object-based image analysis (OBIA).....	159
Table 16.2. Appendix: Expanded Table Table 8.4. Linear models of mean NDVI value and percent live cottonwood canopy per reach.	159

Abstract

Snake Creek flows east from the southern Snake Range in Nevada over complex lithology before leaving Great Basin National Park. The river travels over a section of karst limestone where some surface water naturally recharges the groundwater flow system. In 1961 a water diversion pipeline was constructed by downstream water users to transport surface water through the groundwater recharge zone to reduce potential water losses. The diversion pipeline dewateres a 5-km reach for most of the year by transporting water past the recharge zone then returning it to the channel downstream. Snake Creek was incorporated into the newly established Great Basin National Park in 1986, and today park managers and visitors are concerned that the diversion has destabilized Snake Creek's riparian ecosystem in this arid region where it has high ecological value. The objectives of this study were to 1) document riparian cottonwood forest conditions in the pipeline-dewatered (DW) reach, 2) evaluate Snake Creek water availability and whether it can support a healthy riparian ecosystem, and 3) determine if dewatering has shifted the fluvial system into an unnatural and poorly functioning state.

We pursued these ecohydrological study objectives in 11 research investigations of Snake Creek's DW reach and nearby reference reaches. The research investigations analyzed: 1) riparian forest condition, tree age, growth, and death; 2) tree ring chronologies through time and space; 3) hydroclimatic drivers of tree growth; 4) stable carbon isotopes extracted from tree rings; 5) cottonwood ecophysiology related to water transport and water stress; 6) historical aerial photography; 7) stand-level riparian forest production; 8) groundwater availability as related to surface water and plant rooting zones; 9) near-surface geophysics using electrical resistivity imaging; 10) channel and valley geomorphology; and 11) in-channel wood jams caused by fallen trees. Integrating these diverse research topics provided a full perspective of historical and modern conditions along Snake Creek.

We found that modern hydrological conditions in Snake Creek's DW reach could not maintain the drought-sensitive ecosystem. The riparian cottonwoods (*Populus angustifolia* and *P. angustifolia* x. *P. trichocarpa*) have experienced significant dieback. Tree mortality was 2.4 times higher in the DW reach than in reference reaches, and surviving trees supported only 60% of the live canopy compared to trees in reference reaches. Changes in the DW reach forest began in the 1960s and became more severe during the last two decades. Stable carbon isotope ratios and branch dieback analyses both demonstrated initial forest adjustments related to water stress beginning in the early 1960s. Tree ring width chronologies indicated two periods of growth decline in the DW relative to control reaches. The first decline in the 1960s represented an immediate adjustment to the modified flow regime, and the second decline in the 2000s demonstrated reduced resilience to atmospheric drought. Aerial photos and stand-level forest production calculations indicated that substantial riparian forest decline occurred in the 1990s–2010s in the DW reach compared to reference reaches. Stable carbon isotope ratios and leaf water potentials revealed that trees in the DW reach experienced greater drought stress than those in reference reaches. Monitoring wells and electrical resistivity surveys both showed riparian water tables to be largely supported by in-channel surface water flow, indicating that the flow diversion removed water that recharges alluvial groundwater and sustains riparian plants. Areas

of widespread tree mortality in the DW reach also corresponded to a larger and more unstable channel with a high instream wood load from fallen trees. Modern conditions of Snake Creek in the DW reach robustly suggest that dewatering the river and its associated riparian corridor adversely affected the riparian ecosystem. The degraded condition is likely to persist and intensify unless water is returned to the channel. As we documented during the wet 1980s and the scientific literature suggest, a partial recovery of the riparian ecosystem is likely possible with restored flows.

Acknowledgements

This report details a highly collaborative investigation among several people who deserve recognition. We thank Ben Roberts of Great Basin National Park for his persistent support and guidance. Additional Great Basin National Park staff members supported our efforts, most notably Jonathan Reynolds who collected hydrologic data. NPS Water Resources Division staff Joel Wagner and Mike Martin helped initiate this project, and Joel provided an insightful review of the full report. Thank you to Ellis Margolis and Mike Scott for thoughtful reviews that improved the caliber of this final report. Plant physiologists Troy Ocheltree (Colorado State University), Sean Gleason (USDA Agricultural Research Station), and Dario Zambonini (University of Padova) helped design and interpret the cottonwood ecophysiology experiments. Thanks to Adam Csank (University of Nevada, Reno) and Craig Stricker (USGS) who processed tree ring carbon isotopes in their research labs. Stewart Rood (University of Lethbridge) provided valuable interpretations and review of the ecophysiology analysis. Jack Wood (NPS Geologic Resources Division) helped jumpstart aerial photo analyses. We benefited from field data collection and project strategizing with Lewis Messner, Liangjun “Dream” Zhu, Tae Wan Kim, Andrew Pfeiffer, and Dario Zambonini. This research was funded by the National Park Service's (NPS) Water Resources Division, Great Basin National Park, and a Focused Condition Assessment grant from the NPS's Natural Resource Condition Assessment Program.

1. Introduction

1.1 Background

Streams and their riparian forests develop through interactions among climate, geology, and ecological processes in a watershed (Naiman et al., 2010; Wohl, 2014). Riparian forests are habitats dependent on fluvial processes, making their species vulnerable to changes in stream flow regimes (Diehl et al., 2018; Nilsson & Berggren, 2000; Stromberg & Merritt, 2016). Flow management actions such as damming or water diversions can reduce the functional capacity of river corridors, affecting erosion and sedimentation (Dean et al., 2011; Liro, 2019), biodiversity (Kuiper et al. 2014), and instream and riparian habitats (Gurnell et al., 2012; Latli et al., 2019; Reynolds et al., 2014). Flow manipulations can destabilize rivers and cause a transition from riparian to upland ecosystems (Merritt & Cooper, 2000; Nilsson & Berggren, 2000). Fish and other instream biota can disappear when streams are dewatered, even if it is just some of the flow or during some of the year (Perkin et al., 2015). Close relationships exist between flow and sediment transport, and alterations to either component can cause changes in the other (Wohl et al., 2015). For example, flow reductions can kill the riparian vegetation that captures sediment and stabilizes banks, leading to increased downstream sediment delivery (Diehl et al., 2018; Pollen & Simon, 2005). Regardless of the direction of change, channel aggradation and incision affect functional relationships between rivers and adjacent habitats. In arid regions such as much of the American West, river corridors provide especially valuable ecological habitats along with social and economic benefits to human society.

Large and abrupt changes to river flow regimes can cause immediate and irreversible impacts to the river corridor (Nilsson & Berggren, 2000), while small and slow perturbations are often absorbed by resilient ecosystems (Poff & Zimmerman, 2010). Effects of moderate perturbations, such as partial flow removal, may be more difficult to detect because ecological resilience and time lags can delay detection of changes for decades after a disturbance begins. Time lags present a basic challenge in landscape interpretation because they obscure cause-effect relationships (Chin et al., 2014; Frank et al., 2015; Poepl et al., 2017). For example, climate warming lags behind greenhouse gas emission by several decades. Delayed, diffuse, and compounding effects make managing for time lags challenging. In a second example, the Dust Bowl of the 1930's was a dramatic event that was primed by decades of mostly undetected gradual vegetation and soil degradation. Eventually the resilience of the soil system was overcome during an atmospheric drought. The inherent resilience of a system, be it the atmosphere, the soil, or a riparian forest, is critical to long-term response (Poepl et al., 2017). Diverse retrospective research approaches can build an understanding of these obscured relationships.

Cottonwoods, in the genus *Populus*, are the most common riparian trees across the central and western United States (Friedman et al., 2005). Cottonwoods can become large trees with biological reserves that help buffer them against stressors. However, they are the most sensitive species to drought induced dieback and mortality (Tyree et al., 1994) and die if their water source is removed (Albertson & Weaver, 1945; Tai et al., 2018). The vulnerability of cottonwoods to natural or man-made drought threatens entire riparian ecosystems when changes in water access exceed the trees'

ability to adjust (Scott et al., 2000). Widespread mortality of this keystone species can cascade and destabilize multiple physical and biological processes within a riparian corridor.

Just like riparian ecological communities depend on streamflow, urban centers and agriculture across the American West also rely on rivers and streams. Moving streamflow into water diversion structures has led to immediate economic and societal benefits, and changing human access to that water would require behavioral shifts. However, creative approaches to evaluating social-ecological systems can help satisfy the multiple demands placed on rivers (Albert et al., 2019; Chin et al., 2014; Kattel, 2019). To identify the best solutions to complex water management issues, a holistic understanding of riparian ecosystems and their ecological processes is needed.

1.2 Study site: The Snake Creek drainage basin and water diversion pipeline

Snake Creek in Great Basin National Park (GRBA) flows from the Southern Snake Range in eastern Nevada and drains into a section of Snake Valley that is in western Utah. Its watershed starts on the eastern flanks of Pyramid Peak and Mt. Washington and is the largest in GRBA at 59 km² (Prudic et al., 2015). While in GRBA, the river travels over Pole Canyon limestone before crossing a detachment fault and flowing over an isolated block of Prospect Mountain quartzite (Figure 1.1; Prudic et al., 2015). The Pole Canyon limestone is a porous carbonate that allows some Snake Creek surface water to infiltrate into the groundwater system. The subsurface flow paths of this recharged groundwater are only partially understood. An ongoing NPS-USGS study plans to quantify the amount of this recharged groundwater that re-emerges from two down-valley springs that return to Snake Creek, as well as the amount that remains in the subsurface and replenishes the deep regional or adjacent valley-fill aquifers (Phil Gardner, pers. comm. 28 Oct 2019).

Most Snake Creek flow that exits GRBA is used in agriculture near the town of Garrison, Utah (Prudic et al., 2015). Over a half-century ago, the agricultural water users concluded that the surface water infiltrating the Pole Canyon limestone was unavailable for their beneficial use, so in 1961 a 5-km long diversion pipeline was constructed to bypass the limestone reach to augment downstream water delivery. The diversion captures Snake Creek discharge in a pipeline and transports it through the reach before returning the water to the channel near springs that presumably indicated the end of surface water loss. Water year 2019 measurements indicate that the pipeline diverts up to about 10 cubic feet per second (cfs; 0.3 cubic meters per second (cms)) of Snake Creek flow. Any greater discharge overtops the pipeline intake, remains in the natural channel, and begins flowing through the dewatered reach. However, due to the losing nature of the reach and the unnaturally dry alluvial aquifer after no-flow periods, much of the surface flow infiltrates into the subsurface. After the dry interstitial space in the alluvial aquifer has refilled, repeated 2019 discharge measurements revealed that approximately 3 cfs (0.08 cms) of surface flow recharges the groundwater system in the upper half of the dewatered reach. The lower half of the dewatered reach is hydrologically neutral, lacking substantial surface water gains or losses. Enough runoff to create continuous flow in the dewatered reach is thought to occur approximately one out of three years in this snowmelt-dominated flow regime (GRBA staff, *pers. comm.*). Therefore, the dewatered reach of Snake Creek is not always completely dry; instead the diversion partially dewateres the stream and leaves most of the channel completely dry in some years and seasonally dry in others.

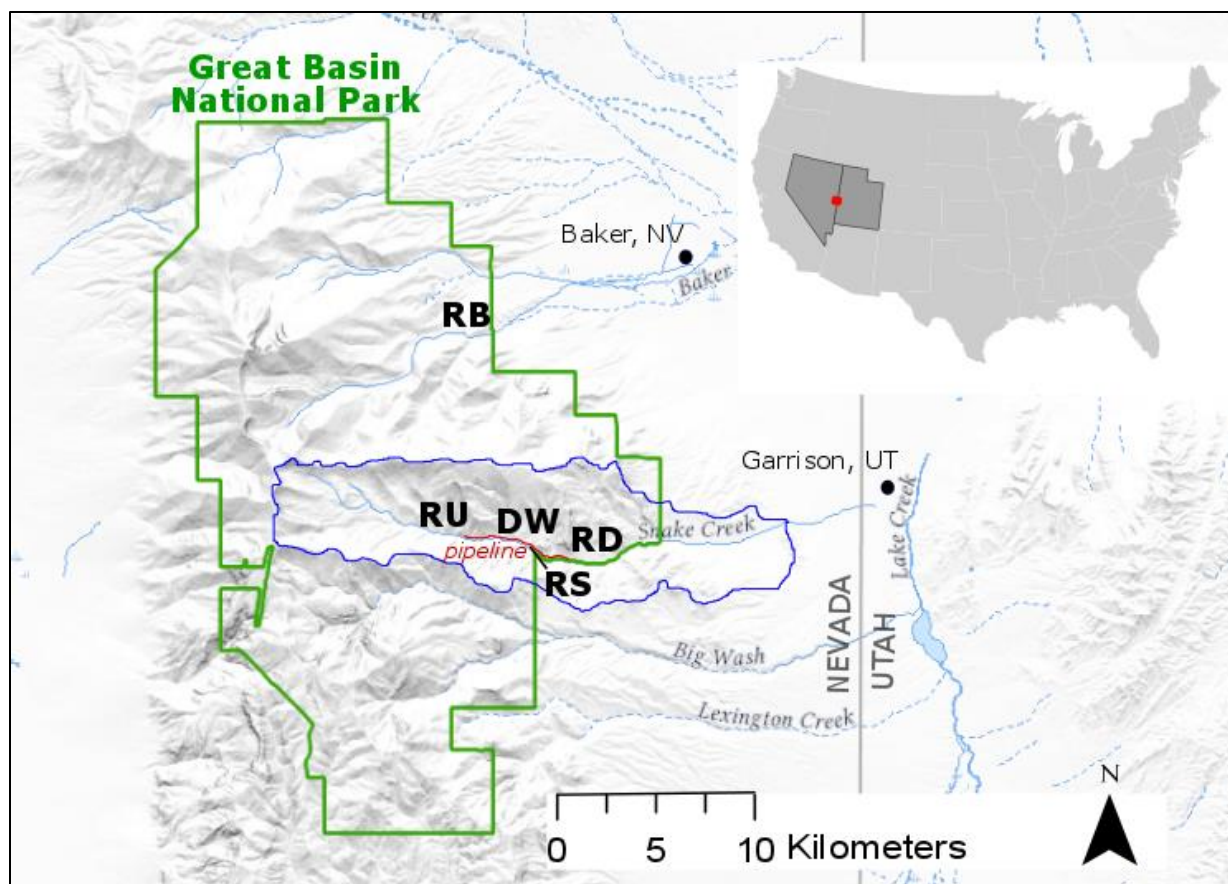


Figure 1.1. Snake Creek and the study reaches in Great Basin National Park: DW = dewatered reach, RU = reference upstream, RD = reference downstream, RB = reference Baker, RS = reference Squirrel. The Snake Creek watershed is outlined in blue, and the pipeline is in red.

National Park Service managers are concerned that the water diversion pipeline has caused channel incision, riparian vegetation dieback, degraded habitat, and a destabilized riparian corridor. Deciduous riparian trees have historically grown throughout most of the Snake Creek riparian corridor within GRBA, including many forest areas dominated by hybrids of narrowleaf and black cottonwood (*Populus angustifolia* x *trichocarpa*). Recently, park managers and visitors have observed changes to the Snake Creek riparian corridor, including areas of dead cottonwood branches and whole trees. It was hypothesized that the tree dieback represents a destabilized ecosystem that is related to the water diversion pipeline. There is further concern that continued operation of the pipeline will prevent forest recovery and lead to further changes to the riparian corridor. Therefore, a more detailed understanding of the conditions and processes occurring in the ecosystem is desired.

1.3 Study approach

This study was designed to identify the role that streamflow plays in supporting the Snake Creek riparian corridor and cottonwood forest. We investigated conditions in the dewatered reach and relationships between physical and biological processes. We hypothesized that if the flow diversion removed water that was vital to the riparian ecosystem, then compared to reference reaches we would find less water, a less stable channel, and more tree dieback. Our objectives were to first document

conditions in the dewatered reach compared to reference conditions, and second to identify the spatial and temporal patterns of changes to better understand their causes.

Our analyses compared Snake Creek's 5-km dewatered (DW) reach to four reference reaches. The upstream reference (RU) reach was immediately above the DW reach, and the downstream reference (RD) reach was immediately below the DW reach (Figure 1.1). The Reference Squirrel (RS) reach was used in tree ring analysis only and was located within the DW reach where local groundwater discharge delivered alternatively sourced water. The Baker Creek reference (RB) reach was at a similar elevation to the DW reach but in an adjacent watershed, where Baker Creek flowed over the same Pole Canyon limestone found in the DW reach. Any reference reach could differ from the affected reach in elevation, watershed area, geology, and other factors. We dealt with this challenge by comparing the DW reach to multiple reference reaches and by investigating all reaches before and after flow diversion in Snake Creek.

This project Introduction is followed by a Study Site description and eleven research investigations that form Sections 3 through 13. Sections 3 through 7 describe the growth and death of riparian cottonwoods, and analyses rely on dendrochronological techniques that leverage information contained within annual growth rings of the trees. Section 8 uses historical aerial imagery as an alternative technique to reconstruct the past, and Section 9 combines datasets to describe riparian forest productivity through time. Sections 10 and 11 characterize near-surface geology and water availability in the Snake Creek riparian corridor, and sections 12 and 13 describe channel morphological characteristics of the Snake Creek channel. A description of these eleven research sections follows.

Cottonwood forest condition: Demography, canopy vigor, and tree mortality. Section 3 uses 2,352 cottonwoods to describe the riparian forest, including tree age structure and the proportion of living canopy of the trees. It also quantifies tree mortality, including analyses of the timing of tree death and branch dieback.

Tree ring chronologies. Section 4 presents tree ring chronologies that show spatial and temporal patterns of annual tree ring growth in 223 cottonwoods. The investigation focuses on growth patterns of DW reach trees over the last century and how they compare to reference trees. Additional comparisons are for live vs. dead trees and trees from different age classes, subreaches, and proportions of living tree canopy.

Stable isotopes in tree rings. Section 5 uses stable carbon isotopes ($\delta^{13}\text{C}$) from 783 annual growth rings in 16 trees to investigate tree water stress through time. The rings are from the DW and reference reaches of Snake Creek and cover the years 1942–2016, characterizing drought stress through space and time. The analysis also compares $\delta^{13}\text{C}$ and tree ring width patterns, showing that diverse metrics within a tree ring can build understanding of growing conditions related to water availability.

Hydroclimatic drivers of tree growth. Section 6 identifies relationships between tree ring width and the possible driving variables of flow, precipitation, and temperature. Relationships are evaluated for

periods before and after the water diversion pipeline was installed, and the results indicate the relative strength of each parameter in supporting cottonwoods across study reaches.

Cottonwood ecophysiology. Section 7 takes a mechanistic view of tree-water relations to characterize physiological processes related to water stress in 20 study trees. Analyses compare cottonwood leaf water potentials over the day and night for trees in the DW and RD reaches. It also includes an experiment showing how effectively trees in the reaches conduct water after being subjected to different levels of water stress.

Aerial photos. Section 8 characterizes the Snake Creek riparian forest from 1946–2017. It uses an object-based image analysis to quantify riparian tree cover through time in 14 photo years. It also uses the normalized difference vegetation index to demonstrate spectral differences through time for trees across reaches and with different proportions of live canopy.

Stand-level riparian forest production. Section 9 scales up cottonwood growth from the tree-level to the stand-level. It uses two approaches that are grounded in our prior analyses to compare riparian forest growth across reaches in terms of annual wood production.

Groundwater monitoring. Section 10 describes information derived from eight monitoring wells and five staff gauges installed along Snake Creek. Well data describe the depth and duration of alluvial groundwater as related to Snake Creek surface water. The section also characterizes geologic material extracted from the boreholes.

Near-surface geophysics. Section 11 describes subsurface conditions around Snake Creek as interpreted using electrical resistivity imaging. Ten transects were used to interpret subsurface conditions, including geologic contacts, saturation levels, and relationships between the channel and alluvial groundwater.

Channel and valley morphology. Section 12 uses 274 channel cross-sections to analyze width, depth, cross-sectional area, and cobble size throughout Snake Creek. It also characterizes valley confinement and slope to assess if landscape setting could cause differences in channel dimensions identified.

Instream wood. Section 13 investigates dead cottonwoods and the role of wood as a geomorphic agent in influencing Snake Creek and its channel dimensions. It characterizes 238 log jams and describes how they vary in frequency and size throughout the length of Snake Creek.

Finally, Section 14 summarizes and synthesizes the research findings.

2. Study Site: Geology, Weather, and River Flow

2.1 Study Reaches

2.1.1 Dewatered (DW) Reach

The DW reach is the 5-km reach of Snake Creek parallel to the water diversion pipeline, beginning at the pipeline inlet and ending at the pipeline outlet (Figure 1.1; Table 2.1). The channel is dry for most of every year (Figure 2.1), but it supports periodic discharge where the pipe leaks and when inflow exceeds the pipeline's intake capacity of about 10 cfs (0.3 cms). Squirrel Springs is located about halfway through the reach and delivers seasonal surface water into the reach. A few additional groundwater discharge areas below Squirrel Springs supply smaller quantities of short-duration flow. The channel passes through varied terrain and geology, and differences in cottonwood growth, channel dimensions, and valley setting by subreach are explored in Sections 4 and 11. Most natural groundwater recharge in the site is thought to occur into the Pole Canyon limestone within the upper 1.6 km (1 mile) of the DW reach (Prudic et al., 2015).

The DW reach riparian corridor includes both wetland and upland trees and shrub species. Deciduous species include cottonwood (*Populus angustifolia* and *P. angustifolia* x *P. trichocarpa*), aspen (*Populus tremuloides*), water birch (*Betula occidentalis*), narrowleaf willow (*Salix exigua*), Wood's rose (*Rosa woodsii*), and chokecherry (*Prunus virginiana*). Evergreen species include singleleaf pinyon pine (*Pinus monophylla*), Utah juniper (*Juniperus osteosperma*), and white fir (*Abies concolor*). In general, in the upper portion of the reach cottonwoods are located along the edges of the stream where they are interspersed with conifers. Around Squirrel Springs the valley widens and there are several areas with a riparian forest 10 to 30 m wide. The physical appearance of the cottonwoods varies throughout the DW reach, with some areas having mostly live and leafy branches, while in other areas trees and branches are mostly or all dead (Figure 2.2). There are active leaks in the pipeline within the reach supporting a more verdant local riparian plant community.



Figure 2.1. Upstream views of Snake Creek in the dewatered (DW) reach.

Table 2.1. Description of the study reaches including physical setting, percent cover for riparian trees and shrubs, and analyses in which each was used. Any modifications to reach extent are noted in specific research sections.

Category	Dewatered (DW)	Reference Upstream (RU)	Reference Downstream (RD)	Reference Baker (RB)	Reference Squirrel (RS)
	4900	780	340	900	
Elevation (m.a.s.l.)	2296–2067	2358–2296	2067–2047	2109–2041 m	2162–2156
#1 species	Cottonwood (32%)	White fir (46%)	Cottonwood (39%)	Pinyon (29%)	Cottonwood (55%)
#2 species	Juniper (18%)	Cottonwood (30%)	River birch (28%)	Cottonwood (22%)	River birch (25%)
#3 species	Pinyon (17%)	Aspen (13%)	Juniper (16%)	Willow (14%)	Pinyon (10%)
#4 species	White fir (8%)	River birch (9%)	Pinyon (12%)	Aspen (14%)	Juniper (5%)
#5 species	River birch (6%)	Juniper (3%)	Sagebrush (3%)	Sagebrush (11%)	Willow (5%)
Analysis: Forest condition	Yes	Yes	Yes	Yes	No
Analysis: Tree ring chronologies	Yes	Yes	Yes	Yes	Yes
Analysis: Isotopes	Yes	Yes	Yes	No	No
Analysis: Hydroclimatic drivers	Yes	Yes	Yes	Yes	Yes
Analysis: Ecophysiology	Yes	No	Yes	No	No
Analysis: Aerial photos	Yes	Yes	Yes	No	No
Analysis: Groundwater wells	Yes	No	Yes	No	No
Analysis: Geophysics	Yes	Yes	Yes	No	No
Analysis: Channel morphology	Yes	Yes	Yes	No	No
Analysis: Instream wood	Yes	Yes	Yes	No	No

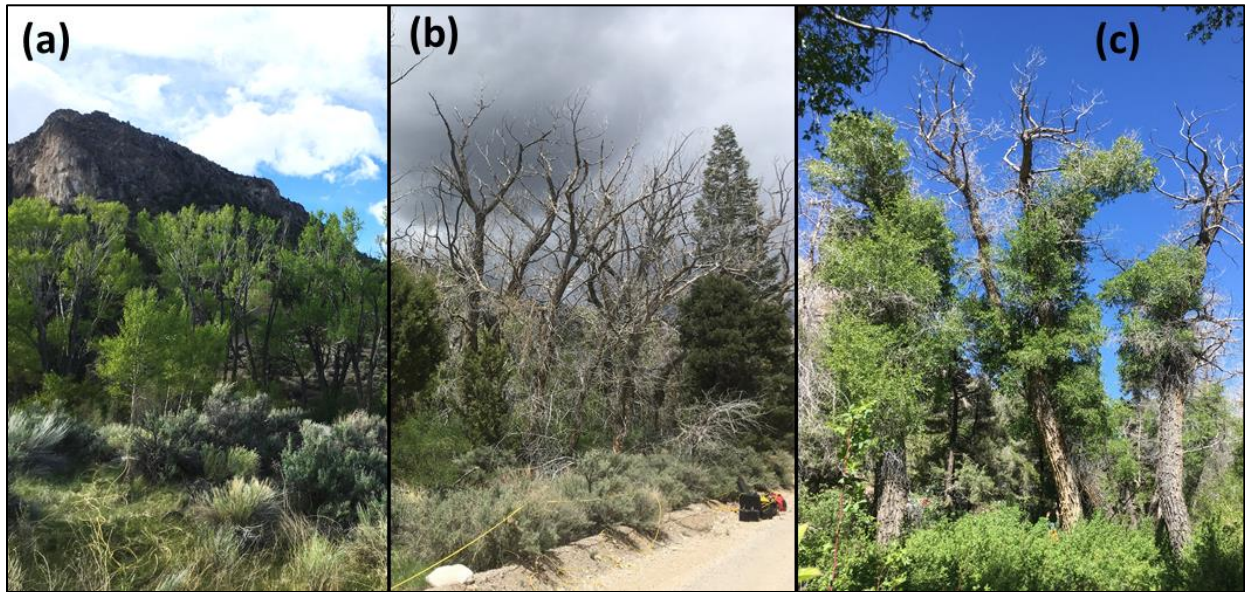


Figure 2.2. Pictures of cottonwood stands found along Snake Creek. (a) Healthy trees in the RD reach, (b) mostly dead trees in the DW reach, and (c) trees in intermediate condition in the DW reach. (c) shows the oldest cottonwoods documented, which had datable tree rings back to 1782 and estimated germination in 1776.

2.1.2 Reference Reaches

Four reference reaches were used in all or some aspects of our investigation of Snake Creek (Table 2.1). The Reference Upstream (RU) reach extended for 780 m immediately upstream from the DW reach and included the Eagle Peak camping area. There was perennial flow in the reach with baseflows of < 1 cfs (0.03 cms). The riparian forest was dense with tall trees, most of which are white firs and cottonwoods.

The Reference Downstream (RD) reach extended for 340 m below the pipeline outlet and had an extensive cottonwood forest with understory vegetation including Wood's rose. There was perennial inflow in the reach from springs emerging near the pipeline outlet and from pipeline outflow. It is unclear to what extent pipeline operation augments flow through this reach relative to conditions before 1961. Given the presence of natural springs, any augmented flow likely affects riparian water table depth by only a small amount.

The Reference Baker (RB) reach was located on Baker Creek, which also drains the eastern flank of the Southern Snake Range in GRBA. The watershed is adjacent to and north of the Snake Creek watershed, and the RB reach was 8 km north of the DW reach. Baker Creek has perennial discharge that drains the south side of Wheeler and Jeff Davis Peaks. The 900-m study reach began just above the Pole Canyon trailhead and extended downstream beyond the park boundary. The elevation of the reach was the same as the lower portions of the DW reach and the RD reach. The primary lithologic unit in the reach is Pole Canyon limestone, which is also found in the DW reach and was the presumed cause of Snake Creek surface water infiltration. Several sparse patches of cottonwoods

were found primarily growing within 5–10 m of the channel, similar to the cottonwood distribution in the upper portions of the DW reach.

The Reference Squirrel (RS) reach was 150 m long and a subreach of the DW reach. It was used as a reference reach because local groundwater discharge, intermittent surface water flow, and electrical resistivity surveys all indicated that trees in RS had access to groundwater. This short subreach was included in the DW reach for most analyses because it was part of the variability of Snake Creek adjacent to the pipeline; however, its trees were used to construct a fourth reference dataset in our tree ring analyses. RS began 500 m below Squirrel Springs where the first cottonwoods appeared after a gap in trees where former Bonita mining activities occurred. The reach experiences seasonal flow from Squirrel Springs and other groundwater discharge. There is a geologic contact between the Pole Canyon Limestone and Prospect Mountain quartzite in this area (McGrew & Miller, 1995), which may cause the groundwater upwelling. The riparian vegetation appeared unusually green and healthy in this reach compared to most areas in the DW reach.

2.2 Geology and Hydrogeology

Geology of the Snake Creek watershed is complex, and secondary factors including geologic structures and karst development are likely dominant factors controlling the movement of surface water and groundwater (Prudic et al., 2015). Geologic units pertinent to Snake Creek's hydrology range from Quaternary alluvium to Precambrian quartzite, with the units of most interest being limestone and dolomite carbonates and quartzite (Table 2.2). The Snake Creek headwaters is largely underlain by granite intruded into the Paleozoic sedimentary section during Jurassic time, approximately 160 million years ago. Most Snake Creek flow begins as snowmelt in the Snake Range below Pyramid Peak (3,635 m) and Mount Washington (3,553 m). The high elevation granite is presumed to be largely impermeable with little flow lost to the subsurface above the study reaches. Snake Creek then becomes losing where it flows out of the granite and into rock units of the lower Paleozoic, particularly the Cambrian Pole Canyon limestone.

While the primary lithology of the units within the study area has been used in the designation of whether they behave as aquifer or confining units (McGrew & Miller, 1995), field observations revealed that faulting and fracturing enable some confining units to transmit groundwater. The dominant structural feature in the study area is a large detachment fault known as the Southern Snake Range Detachment. This low-angle fault produced an upper plate and a lower plate. The lower plate is largely composed of the Cambrian Pole Canyon limestone and underlying Cambrian/Precambrian Prospect Mountain quartzite (Cpc and CZpm, respectively, in Figure 2.3). These units were ductilely deformed and partially metamorphosed, especially where the Pole Canyon limestone is near the fault plane or the granitic intrusion. In contrast, the upper plate of the detachment fault exhibits brittle deformation resulting in a heavily faulted and tilted sequence of largely lower Paleozoic carbonates and quartzite. Figure 2.3 shows a cross-section of the study area with the granitic intrusion in the headwaters, the transition into the layered Paleozoic section, the trace of the low-angle detachment fault, and resulting upper and lower plates.

Karst development is another secondary geologic factor affecting hydrogeologic properties of the Snake Creek watershed. Karst develops when limestone or dolomite undergo preferential chemical

and physical weathering and dissolution. While limestone and dolomite can have low primary porosity, karst processes can create large secondary porosities with the capacity to transport large quantities of water. Large-scale karst development is seen as caves in the cliffs above Snake Creek, and nearby at Lehman Caves. Smaller-scale karst development in the Snake Creek watershed is thought to play a significant role in surface water recharge of groundwater, preferential groundwater flows, and spring development.

Table 2.2. Selected geologic units mapped within the study area. Hydrostratigraphic unit types are from Prudic et al. (2015), and unit descriptions are from McGrew and Miller (1995).

Unit Name	Hydro-stratigraphic Unit	Unit Description
Alluvium	Local aquifer	Generally unconsolidated sands and gravels, with cobbles and boulders
Jurassic granite	Impermeable	Granite of the Snake Creek-Williams Canyon pluton, dated to 160My
Fish Haven / Laketown dolomites	LCAU ¹	Undifferentiated brown dolomite with fossil beds and cherty intervals
Eureka quartzite	LCAU	Cliff-forming thick-bedded quartzite; distinct stratigraphic marker
Pogonip Group	LCAU	Yellow-weathering, blue-gray fossiliferous silty limestone.
Notch Peak limestone	LCAU	Medium-gray cliff-forming limestone containing abundant chert
Pole Canyon limestone	LCAU	Blue-gray cliff-forming limestone, partially metamorphosed to low-grade marble in places
Pioche shale	LSCU ²	Siltstone, sandy siltstone, and sandstone that has been metamorphosed to quartzite and schist. Shale rare in map area
Prospect Mountain quartzite	LSCU	White to gray quartzite, weathers brown, tan, or purple. Abundant cross-beds; cliff-former

¹ LCAU: Lower Carbonate Aquifer Unit; ² LSCU: Lower Siliciclastic Confining Unit

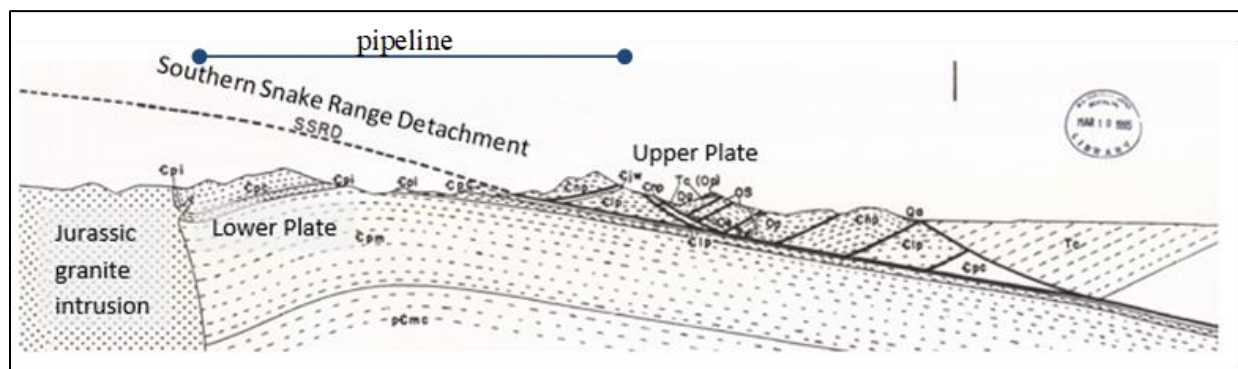


Figure 2.3. Southern Snake Range geologic cross-section showing the transition from the granitic intrusion to layered Paleozoic section downstream, at right. The trace of the detachment fault and the

units and orientation of the upper and lower plate rocks are shown, as is the approximate extent of the pipeline. Modified from McGrew and Miller (1995).

2.3 Weather and Climate

It is well-established that temperature and vapor pressure deficit (VPD) have increased in recent decades across the region (Ficklin & Novick, 2017), including at GRBA (Monahan & Fisichelli, 2014). Extreme weather events or a slowly changing climate are known to instigate ecological changes (Pecl et al., 2017), including tree branch dieback and tree mortality (Rood et al., 2000; Sperry & Love, 2015). Atmospheric drought can be naturally mitigated in riparian ecosystems, however, when streams provide biologically-accessible water.

Spatial patterns of temperature and precipitation in GRBA resemble those from the larger Great Basin physiographic province, where low-lying basins are hot and dry and high-elevation mountain ranges are relatively cool and wet. Around GRBA, mean annual precipitation ranges from 16 cm/yr in the Snake Valley to 81 cm/yr on the highest peaks in the park (Prudic et al., 2015). The National Weather Service (NWS) Coop station at Great Basin National Park, NV (GHCND: USC00263340, USC00264514) is at 2080 m elevation, which is the elevation of the Snake Creek DW reach. This station had a 1981–2010 mean of 35 cm/yr of precipitation that was well distributed throughout the year (Figure 2.4). Mean daily temperatures range from lows of -7°C (19°F) in December and January to highs of 29°C (84°F) in July.

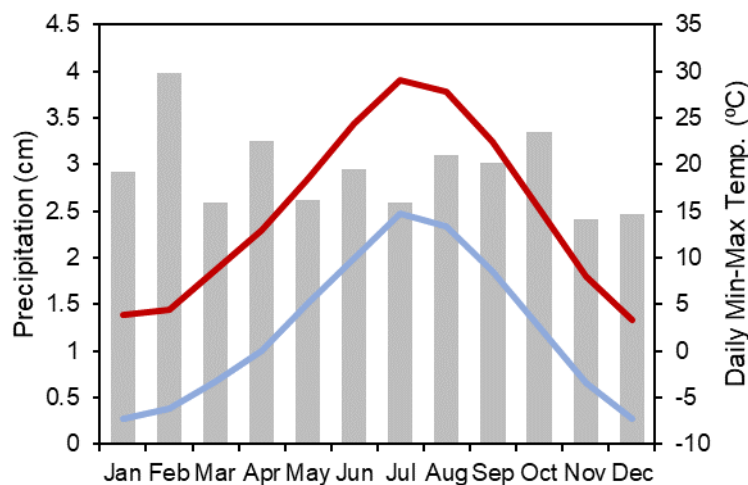


Figure 2.4. Climograph of 1981–2010 monthly average precipitation, daily maximum temperature, and daily minimum temperature at the Great Basin National Park Weather Station.

We characterized climate using PRISM, a gridded dataset that interpolates multiple weather stations using a weighted regression that factors in landscape position (Daly et al., 2008). The benefits of PRISM compared to the NWS Coop station were its longer and complete record and the inclusion of VPD. PRISM data were analyzed for 1910–2015 for a 4 km grid cell in the Snake Creek drainage just upstream from the DW reach (38.93°N , 114.20°W ; 2445 masl). VPD was a derived variable

that is often considered a superior metric of plant water stress because it characterizes the strain under which plants maintain their internal water balance (Anderson, 1936; Williams et al. 2013).

We identified three major droughts in the record, occurring in the 1930s, the 1950–60s, and the 2000s. Each drought was characterized by several years of high temperature and VPD, and the first two droughts also had prolonged periods of low precipitation (Figure 2.5). The very high temperatures in the 2000s drought exacerbated its moderately low precipitation totals, resulting in a “hot drought” also identified in the adjacent Colorado River basin (Udall & Overpeck, 2017). The five highest minimum temperature years all occurred since 2000. The most recent year in the dataset, 2015, had the warmest average minimum temperature at 2.7° C (37° F), compared to the long-term average of 0.6° C (33° F). 2001, 2002, and 2003 were three of the four highest years for both minimum and maximum VPD, indicating that this period would cause atmospheric water stress to plants. When looking only at VPD for the summer growing season from June–August, each of the top ten driest years occurred since 2000. People living through any of these three droughts are likely to have observed environmental stresses that could have inspired action. For example, drought in the 1950–60s could have motivated irrigators to construct the Snake Creek water diversion pipeline, and the 2000s drought may have initiated environmental changes that led to this study.

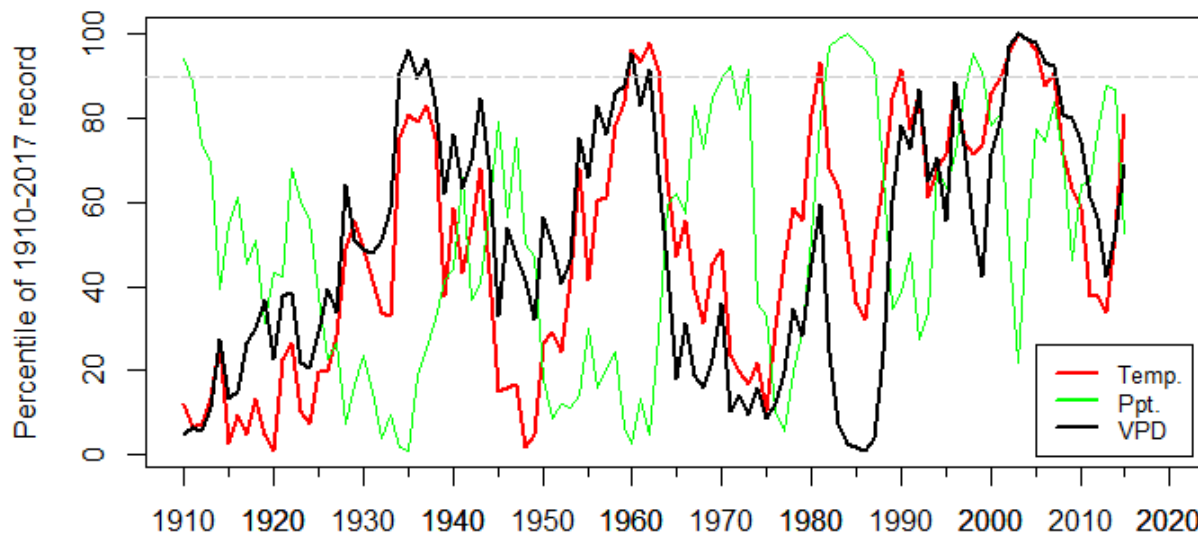


Figure 2.5. Mean temperature, total precipitation, and vapor pressure deficit from PRISM displayed as 5-year running averages. Values are shown as percentiles, where the highest year is 100%, the median year is 50%, and the highest 10% of values are above the dashed grey line. Note that precipitation peaked in the 1980s, while temperature and vapor pressure deficit (VPD) peaked in the 2000s.

2.4 Streamflow

The Snake Creek flow record was short and unusable for long-term analyses, such as linking flow to cottonwood growth. Mean annual flow at NPS gauging station SNK3, located just above the pipeline inlet in GRBA, averaged 4.4 cfs (0.12 cms) over the incomplete measurement record from 2011–2017. We used the longer flow record from Cleve Creek USGS station 10343700 as a proxy for Snake Creek flow, following Prudic et al. (2015). Cleve Creek drains the eastern side of the Schell

Creek Range, 45 km northwest and in a similar landscape position as the also east-flowing Snake Creek. We tested the consistency between the two flow records using daily discharge data from the overlapping period 2011–2017. Snake Creek 2011–2016 water year data were downloaded from the NPS AQUARIUS web portal. Water year 2017 data were provisional, and we calculated 2017 discharge from stage data with high confidence using a stage-discharge relationship from a 3rd degree polynomial constructed from nine field measurements that spanned most of the annual discharge range ($R^2 = 0.998$). Discharge values from the two streams were retained for days common to both records before being averaged into monthly means (Figure 2.6). The correlation between monthly flows in the two streams was moderately high ($R^2 = 0.80$), so Cleve Creek was considered a good proxy dataset for Snake Creek. However, its use likely diluted the signal between flow and cottonwood growth at Snake Creek, so relationships between tree ring growth and flow were likely stronger than those identified (Section 6). Cleve Creek annual peaks tended to be a month earlier than those at Snake Creek (Figure 2.6) but the interannual variation in flow was strongly consistent at these two gauges.

Cleve Creek flow data were used as a proxy for Snake Creek and Baker Creek flows for 1960–2017. Cleve Creek data spanned this period, but 22% of the record was missing. To infill the missing months, we used monthly regression equations developed from 40 years of measurement overlap with neighboring Steptoe Creek. Cleve Creek and Steptoe Creek occupy adjacent watersheds in the Schell Creek Range, just 30 km across the Spring Valley from the Snake Range and GRBA. Their monthly flow correlations were high ($0.75 < R^2 < 0.95$), with higher correlations during the high runoff months that most influence the riparian corridor. The annual runoff pattern was characteristic of snowmelt-dominated flow regimes, with peak annual flows typically occurring in May or June and baseflows being punctuated by occasional storm runoff throughout the non-snowmelt season (Figure 2.7; Figure 2.8).

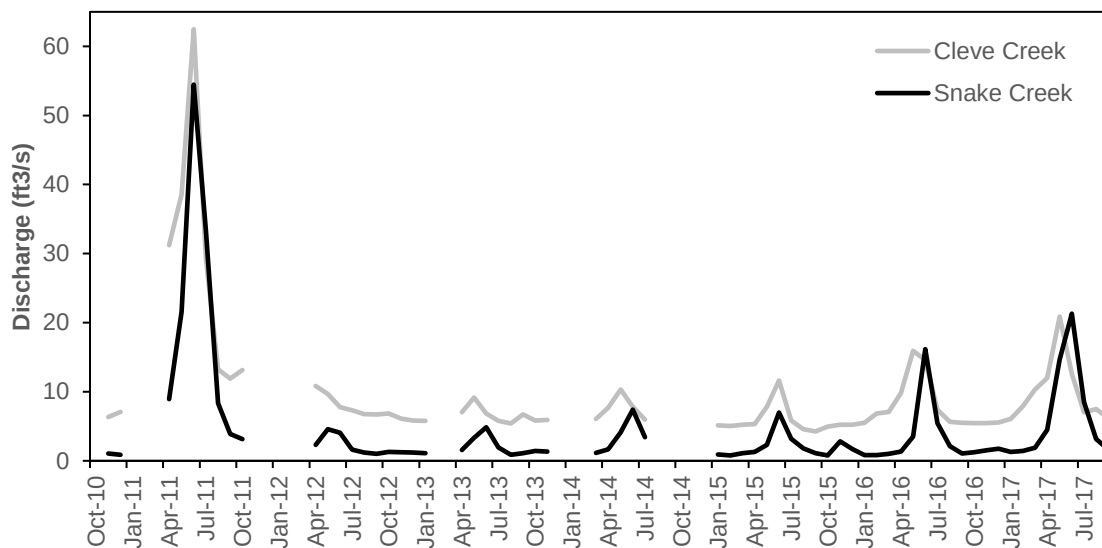


Figure 2.6. Snake Creek and Cleve Creek hydrographs for the period of overlap, water years 2011–2017. Data are mean monthly flow for months with at least 8 common days of data.

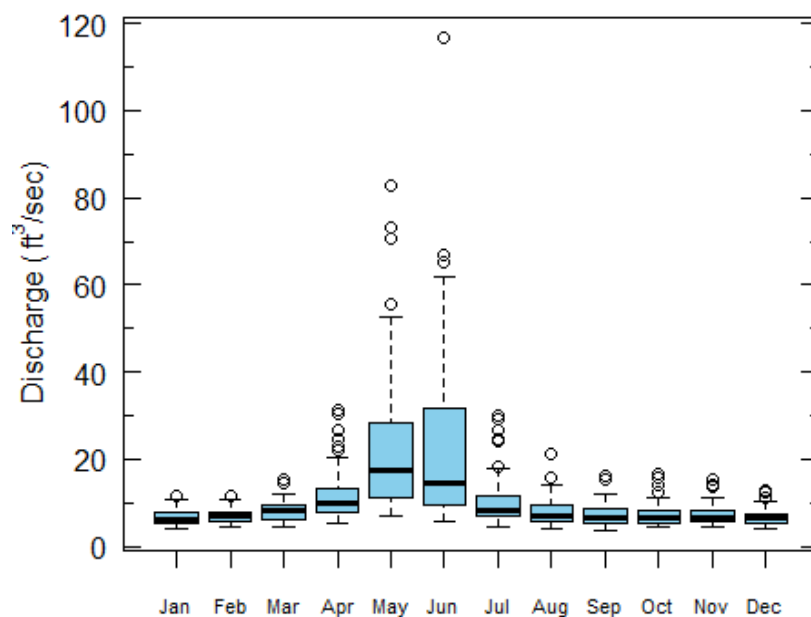


Figure 2.7. Boxplot of the distribution of mean monthly flows at Cleve Creek, water years 1960–2017.

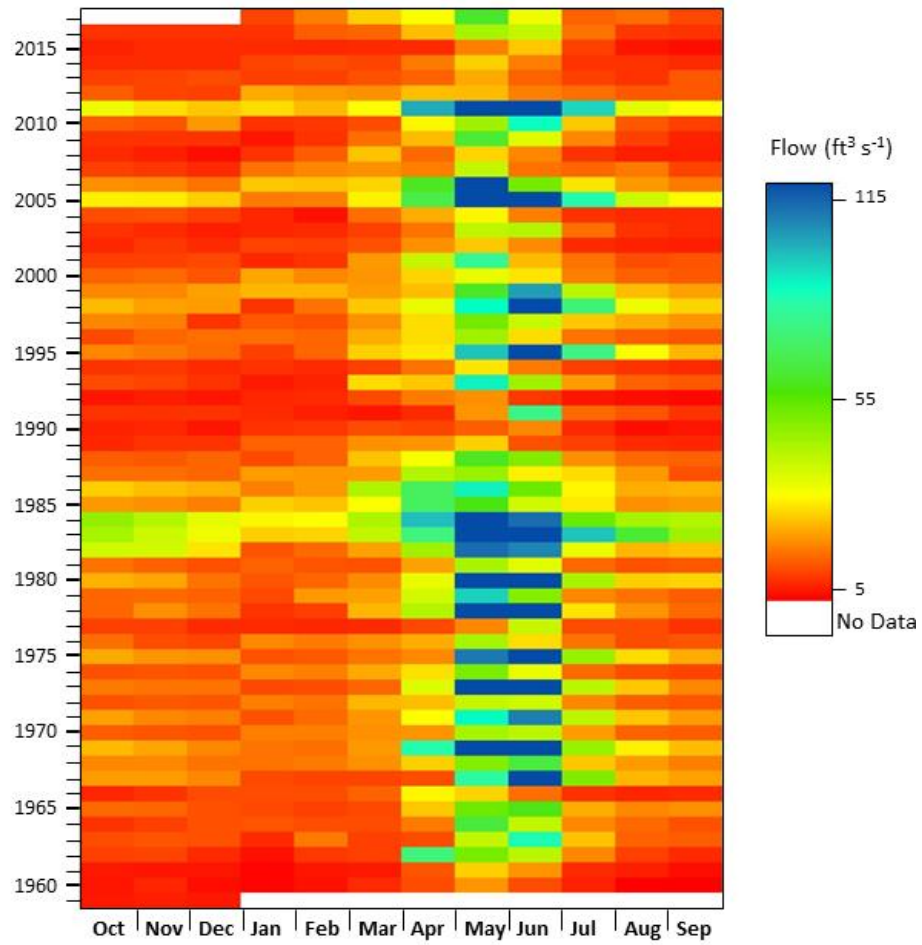


Figure 2.8. Mean monthly flow for the infilled Cleve Creek dataset for water years 1960–2017. Colors are scaled to flow, with red being low and blue high.

3. Cottonwood Forest Condition: Demography, Canopy Vigor, and Tree Mortality

3.1 Introduction

The spatial and temporal patterns of cottonwood vigor can reveal if modern growing conditions are likely to maintain the existing riparian forest. Riparian cottonwood growth occurs with hydrologic connection to rivers and streams, and branches and roots develop in relation to the prevailing hydrologic conditions (Lytle & Poff, 2004). If water is removed, trees may sacrifice their branches to promote survival (Cooper et al., 2003; Rood et al., 2000; Schook et al., 2016a; Williams & Cooper, 2005). This canopy reduction lowers metabolic demands and improves the tree-water balance by reducing leaf area. Reach-level characterization of canopy declines can reveal if unfavorable changes in the growing environment affect a reach. Canopy surveys can be used to document the extent of dieback (Cooper et al., 2003; Rood et al., 2000), and dissecting trees by cross-dating their branches can reveal when and how quickly the sequence of dieback unfolded (Schook et al., 2016a). These structural adjustments can help trees reach a new equilibrium that promotes survival (Lytle & Poff, 2004), or the changes may be unsuccessful and precede tree death. Cottonwood survival after alluvial groundwater access is modified varies by the degree of the modification, and documenting the spatial and temporal patterns of mortality can help identify the causes of spatially isolated tree mortality and survival (Cooper et al., 2003; Scott et al., 2000). We characterized canopy dieback and tree death in GRBA cottonwoods to investigate if hydrologic modifications to Snake Creek's DW reach were related to its riparian forest decline.

3.2 Cottonwood Demography

3.2.1 Survey of Live and Dead Trees

3.2.1.1 Methods:

Between September 2016 and May 2018 every living cottonwood at least 20 cm diameter at breast height (DBH) was surveyed in the dewatered (DW) reach and three main reference reaches (RU, RD, and RB). Dead trees were included if they were free-standing to at least 2 m tall. In May 2018 we resurveyed to document all fallen dead trees recognizable as cottonwoods at least 20 cm diameter. The fallen tree survey was limited to the DW reach because of time constraints. Each tree's DBH was grouped into diameter bins of 20.3 – 40.6, 40.6 – 61.0, 61.0 – 81.3, 81.3 – 101.6, and 101.6 – 121.9 cm (8–16, 16–24, 24–32, 32–40, 40–48 in), numbers that are rounded to whole numbers in the following text. GPS coordinates were taken for each tree using Trimble GeoXT and Geo XH units with a minimum of 4 satellites.

3.2.1.2 Results:

We surveyed 2352 live and dead standing cottonwoods in the four study reaches. Tree density and condition varied within and across reaches. DW was the longest reach and had the most trees ($n = 1581$), and it had the highest proportion of dead trees (28.4%). There were 310 trees in RU (11.0% dead), 262 trees in RD (12.2% dead), and 199 trees in RB (13.1% dead; Table 3.1).

Cottonwood counts in the DW reach were 1132 standing live, 449 standing dead, and 454 fallen dead. The total of 903 standing and fallen dead trees represented 44.4% of all identifiable cottonwoods in the reach. Although fallen dead trees were not surveyed at the reference reaches, we anecdotally observed very few fallen dead trees compared to their prevalence in the DW reach.

Table 3.1. Live or dead status of standing trees in the four primary study reaches.

Category	DW	RU	RD	RB
Live Trees	1132	276	230	173
Dead Trees	449	34	32	26
Total Trees	1581	310	262	199
Percent dead	28.4	11.0	12.2	13.1

3.2.2 Cottonwood Size and Age Distributions

3.2.2.1 Methods

We used two approaches to investigate cottonwood age distribution and tree regeneration. First we broadly assessed the size distribution of tree basal diameters from a canopy survey ($n = 1515$), and second we precisely aged every cored tree ($n = 202$; Figure 3.1).

For the tree diameter assessment, we characterized all cottonwoods at least 5 cm (2 in) DBH in the DW reach into size classes of 5–13, 13–20, 20–41, 41–61, 61–81, 81–102, and 102–122 cm. This required surveying the 5–20 cm trees to supplement the original survey of trees > 20 cm DBH. We standardized tree counts by bin widths in centimeters to compare the number of trees across classes. Cottonwood floodplain forests in equilibrium have been suggested to have a coarsely negative exponential age class distribution (Merigliano et al., 2013), meaning fewer trees would be present in each older age class. This is because younger trees are more densely spaced and have higher mortality rates than older trees. The strongly declining age distribution predicted by the exponential models from unconfined floodplains (Everitt, 1968; Merigliano et al., 2013) may be more skewed than that expected in a confined setting such as the Snake Creek. Regardless, we expected fewer older trees because stands of young trees are more densely spaced. In this analysis we used tree diameter as a proxy for age, where increasing diameters were considered to represent increasingly older trees. The strength of this assumption is also analyzed.

The more precise approach for investigating tree age structure was to age the randomly selected live trees that were cored and dead trees that were wedged. See Section 4.2 for a more detailed description of tree ring methods, including how sampled trees were selected. In summary 12 trees > 20 cm DBH were cored from each live tree canopy class, and a proportional subset matching the distribution of canopy classes was used for the site chronology. Aging trees also enabled testing of the assumption that age and diameter are related. This analysis used trees only from the DW reach because the diameter-age assessment occurred for this reach only. We determined tree ages using standard methods of cross-dating (Stokes & Smiley, 1968). If a sample did not contain pith, which is

formed in the first year of growth, we used the curvature of the rings to calculate the number of missing rings following methods from Meko et al. (2015). Pith dates identified likely lagged germination by 5–10 years because samples were collected 1.3 m above ground, above the germination point (Metzger et al., 2019; Schook et al., 2017; Scott et al., 1997).

We analyzed relationships between tree age and tree diameter in two ways. The first was to find the average annual growth increment for a tree ring for each tree age. This was an adaptation of the Regional Curve Standardization approach, which combines rings from many trees to describe site-level growth rates (Briffa & Melvin, 2011). The approach calculates the mean growth increment for all tree rings of a given age using all trees in the sample. The second approach was simpler and assessed the age-diameter relationship for all cored DW trees using DBH measured for each tree using a circumference tape. The age-diameter relationship was then determined by fitting a power function to the points, which was used because radial expansion slows as tree diameter increases (Biondi & Qeadan, 2008).

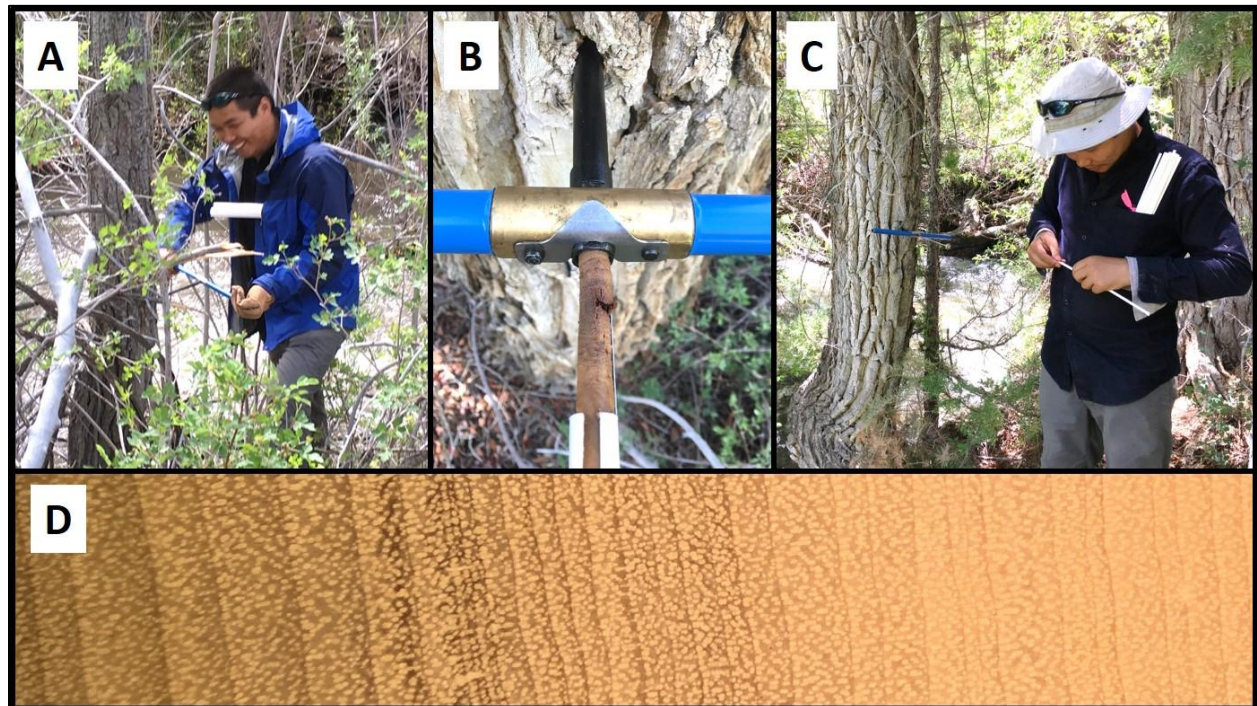


Figure 3.1. Tree core collection included (A) coring, (B) extracting the core from the tree, (C) storing the core in a labeled straw, and (D) analyzing the sanded core under a microscope.

3.2.2.2 Results and Discussion

3.2.2.2.1 Dewatered (DW) Reach Tree Age Structure

Small cottonwoods were rarer than expected in the Snake Creek DW reach, suggesting that young trees may be too rare to replace older trees as they die. The density of small trees was much less than expected for a stable population. There were 38 trees per centimeter of bin width in the 5–13 cm bin, and 16 trees per cm in the 13–20 cm bin. These numbers were not, but were expected to be,

exponentially greater than counts in the larger classes (Table 3.2). For each progressively larger size class beyond 20–41 cm DBH the number of trees decreased as expected. Similar underrepresentation of young trees has been documented on other western rivers (Merigliano et al., 2013; Schook et al., 2016b), and these studies attributed the cottonwood recruitment decline primarily to flow regulation and reduced channel migration.

Cross-dating the cored trees showed tree age to increase for each larger size class, increasing from 100 years to 133, 136, and 202 years for size classes 20–41, 41–61, 61–81, and 81–102 cm DBH, respectively ($n = 72, 32, 9,$ and 1 trees; Figure 3.2; Table 3.2). Cross-dated cores showed that larger-diameter trees were older than smaller trees, but not as much older as would be the case if all trees grew at the same rate (Figure 3.3). For any given tree, its diameter only increases with age. However, each tree’s unique growing conditions create variation in this relationship and tree-specific growing conditions determine annual growth increment and subsequent tree diameter. Based on bark morphology and tree growth patterns, we suspect that many small trees in the 5–13 and 13–20 cm classes were over 50 years old instead of being very young suggesting recent regeneration. This conclusion was supported by analyzing only the 20–25 cm DBH trees which averaged 71 years old. Thus, even though we did not age the small trees, their rarity and slow growth bias provide further evidence that Snake Creek cottonwood regeneration is low. It is both older and faster growing trees in the larger size classes (Brienen et al., 2012). By similar logic, both young and slow growing trees are likely in the small size classes (Brienen et al., 2012). Thus, because slow growing, not just young, trees are represented in the small classes, we conclude that even fewer young trees are present at Snake Creek than suggested by this size class distribution, and recent cottonwood regeneration has been even lower than Figure 3.2 suggests.

Table 3.2. Live dewatered reach trees in each diameter size class. Values are total and number of trees standardized per centimeter of bin width in each size class. DBH = Diameter at breast height.

DBH (cm)	Live DW trees	Trees per cm diameter	Cored Trees (n)	Cored tree age (mean)
5–13	287	38	0	n/a
13–20	124	16	1	59
20–41	633	31	72	100
41–61	367	18	32	133
61–81	98	5	9	136
81–102	5	0.2	1	202
102–122	1	<0.1	0	n/a

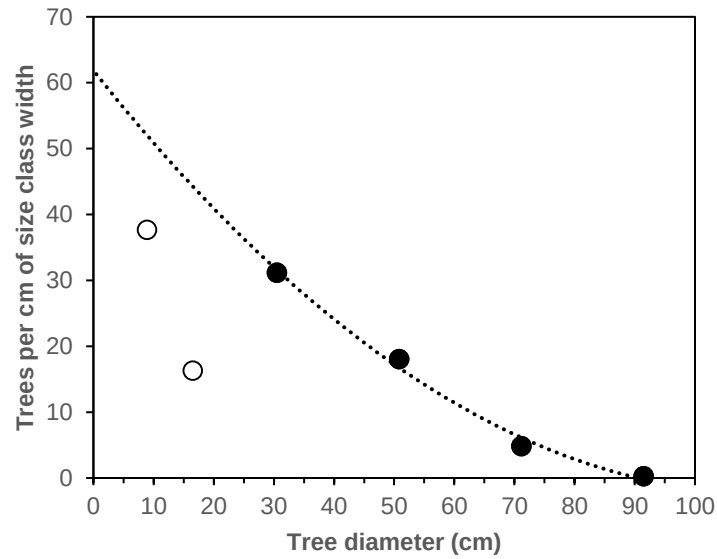


Figure 3.2. Number of trees by tree diameter for each size class in the DW reach, with values scaled to number per centimeter of bin width for each class. Black dotted line is a power function fit to the trees in the four largest size classes (filled circles), showing an underrepresentation of small trees (open circles). Note that the 102–122 cm size bin is excluded (Table 3.2) because no randomly selected and aged trees were this size.

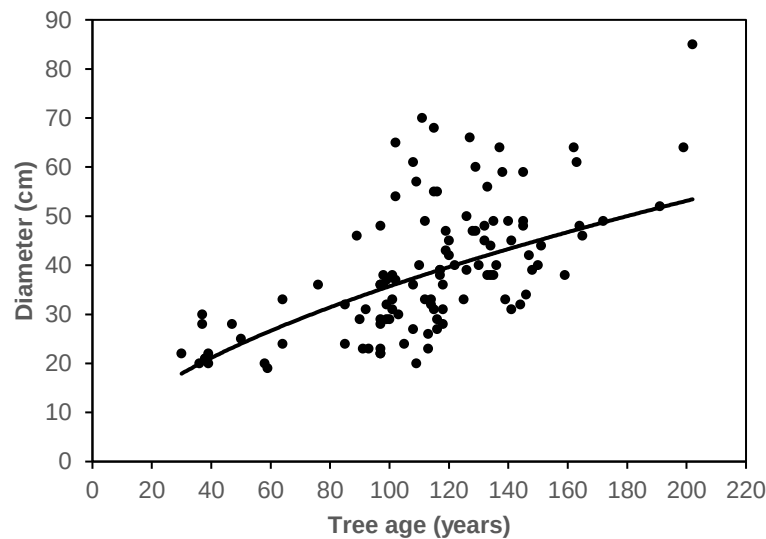


Figure 3.3. Age-Diameter relationship for all live trees sampled in the DW reach in 2017. Power function trendline is fit to the data because it matches the conceptual model of reduced radial expansion with increasing age ($R^2 = 0.42$), but substantial deviation from this model demonstrates natural variability in diameter due to other variables (e.g., water access, nutrient access, shading).

3.2.2.2.2 Tree age structure by reach

The first year of growth was identified for the 202 randomly selected and cored live trees among the study reaches, from which we derived two main findings. The first was that most (70%) of the establishment of cottonwoods at the three Snake Creek study reaches (DW, RU, RD) was between 1870 and 1920 (Figure 3.4). The second key finding is that there were far fewer young trees at Snake Creek than predicted by the exponential model of stable cottonwood populations (Everitt, 1968). 81% of Snake Creek trees established by 1920, and 90% established by 1936. The paucity of young trees indicates that decreased cottonwood regeneration began by 1930. Some combination of climate, land management, and water management likely contributed to the low cottonwood regeneration in recent decades.

Of the three Snake Creek study reaches, cottonwood establishment was the most widespread through time in the DW reach, but still 72% of DW trees dated to the half-century between 1870 and 1920 and 93% established by 1932 (Figure 3.4). The RU reach was a relatively homogenous reach 780 m long, and nearly all of its cottonwoods established synchronously, with 88% of establishment between 1875 and 1899. Similarly but to a lesser degree, at RD 61% of trees established between 1907 and 1927, and only 1 of 28 trees established after 1952.

It could be argued that because cored trees were all >20 cm DBH, very young trees would not be expected in the sample. Given the relationship reported above for tree diameter by age, we would have expected 50% of the trees dating to 1966 to have met the diameter required for sampling. However, virtually no trees were documented to have established around this time. Therefore, the reduced recruitment occurred decades earlier, and the underrepresentation of young cottonwoods was not an artifact of the sampling methodology.

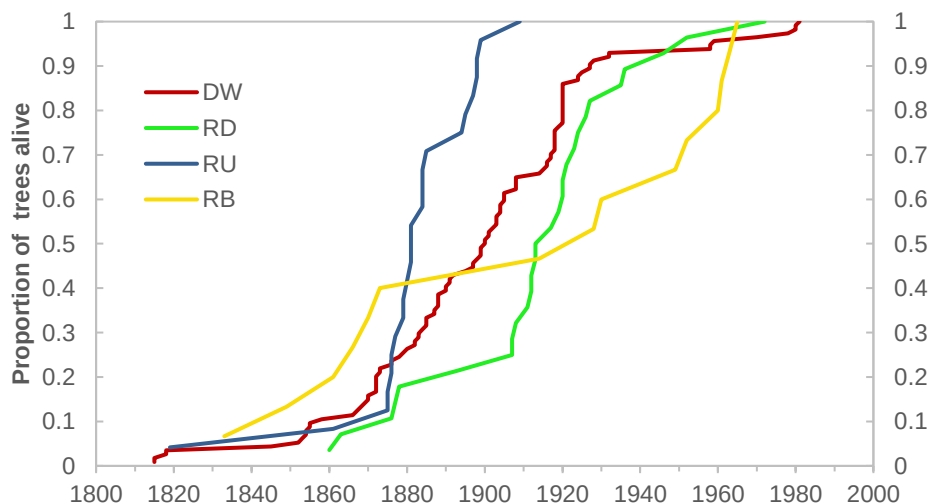


Figure 3.4. Proportion of cored trees establishing by year. Steep portions of the lines represent high recruitment periods. Note that very little establishment occurred after 1930.

3.3 Cottonwood Canopy Vigor

3.3.1 Methods

We classified canopy vigor for all 2352 standing cottonwoods >20 cm DBH in the study reaches using methods adapted from the US Forest Service Crown Condition Classification assessment, which aims to assess the morphological impact of crown stressors (Schomaker et al., 2007). The live branches in the crown area were evaluated between the lowest and the highest branches, and missing portions were not included in the canopy estimates. Trees were binned into groups of 0%, 1–20%, 21–40%, 41–60%, 61–80%, and 81–100% live canopy, which we generally report as 0%, 10%, 30%, 50%, 70%, and 90% live canopy, respectively (Figure 3.5). Canopy classification was calibrated among the three researchers through periodic discussions and convergence to the same value for calibration trees. We compared canopy vigor in the DW reach to the three reference reaches using ANOVA with post-hoc t-test to investigate differences from reference conditions.

Low canopy vigor can be caused by age-related factors instead of environmental factors. To investigate canopy vigor across riparian cottonwoods, we used the 115 aged living DW trees that were selected from the stratified random sample (Section 4.1.1). We separated them into three age classes grouped by their first year of growth to find the average canopy vigor by age class for trees establishing from 1815–1889, 1890–1928, 1932–1987. We used the Kruskal-Wallis test, the non-parametric analog to an ANOVA, with follow up Dunn’s test for multiple comparisons to document differences in the central tendency between groups.

The canopy vigor classification was subjectively assigned, so we validated the six canopy classes by conducting an aerial photo NDVI analysis (Section 8.3.2) and quantifying tree leaf area index (LAI). We calculated LAI for 25–31 trees per canopy class using the AccuPAR LP-80 ceptometer (Decagon Devices, Inc., Pullman, WA). The ceptometer measured photosynthetically active radiation (PAR; i.e., light intensity) under a tree canopy compared to areas in full sun outside of the canopy. Three-to-five measurements were taken outside of the canopy and seven-to-ten were taken below the canopy. Below-canopy measurements were taken from representative locations, instead of from a rigid sampling scheme because of canopy shape, below-canopy access, and interference from surrounding vegetation. Measurements were taken within two hours of solar noon. Calculations incorporated solar zenith angle, sun beam fraction, and leaf angle distribution to calculate LAI (Leblanc et al., 2002). Although LAI measurements were only taken from select trees in the DW and RD reaches, the analysis applies to all reaches because tree canopies were classified the same across reaches. Due to non-normally distributed data, we used a Kruskal-Wallis test with Dunn’s multiple comparison correction factor to identify significant differences in central tendency among groups.

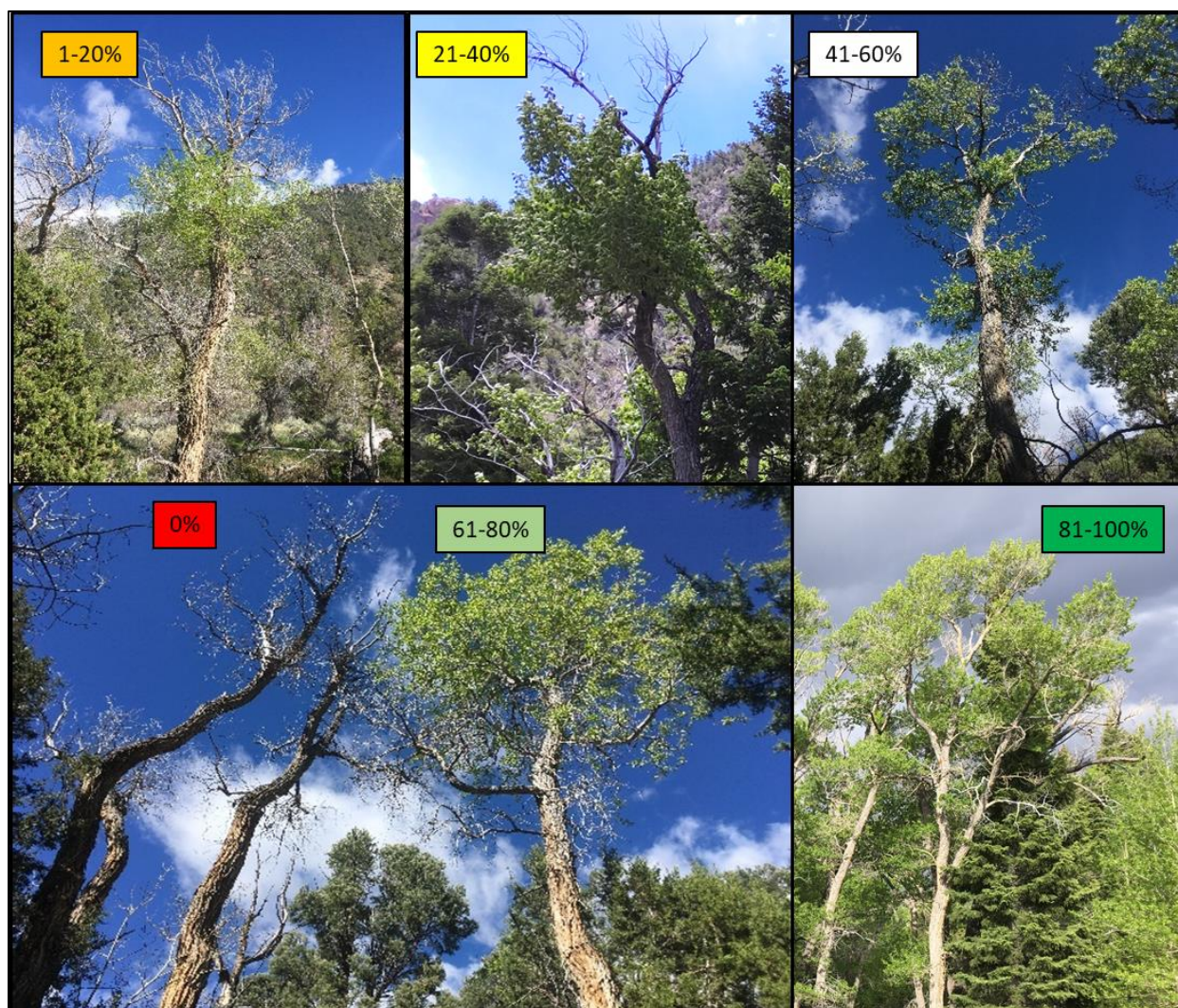


Figure 3.5. Example pictures of cottonwoods in each canopy class: 0%, 1–20%, 21–40%, 41–60%, 61–80%, 81–100% (i.e., 0%, 10%, 30%, 50%, 70%, and 90%, respectively).

3.3.2 Results and Discussion

There were 449 standing dead trees in the DW reach, which was 28% of DW trees and 2.2–2.6x greater than the 11–13% mortality in reference reaches. Mean live canopy for standing DW trees was 30%, compared to 55–67% live canopy for the reference reaches (Figure 3.6a). Cottonwoods in the DW reach had significantly less live canopy than any reference reach (ANOVA test, $F = 154.6$, $p < 0.001$). For live trees only, DW had 43% canopy coverage compared to 63–76% for reference reaches. The reference reaches were similar to the 64–69% live canopy in *P. fremontii* along the Yampa River (Williams & Cooper, 2005). The DW reach had more trees in the 0% and 10% classes than in other classes, while the three reference reaches were opposite having 90% and 70% as the most common canopy classes (Figure 3.6b). The consistency among reference reaches and their difference from DW strongly indicates that the DW reach has declined in vigor.

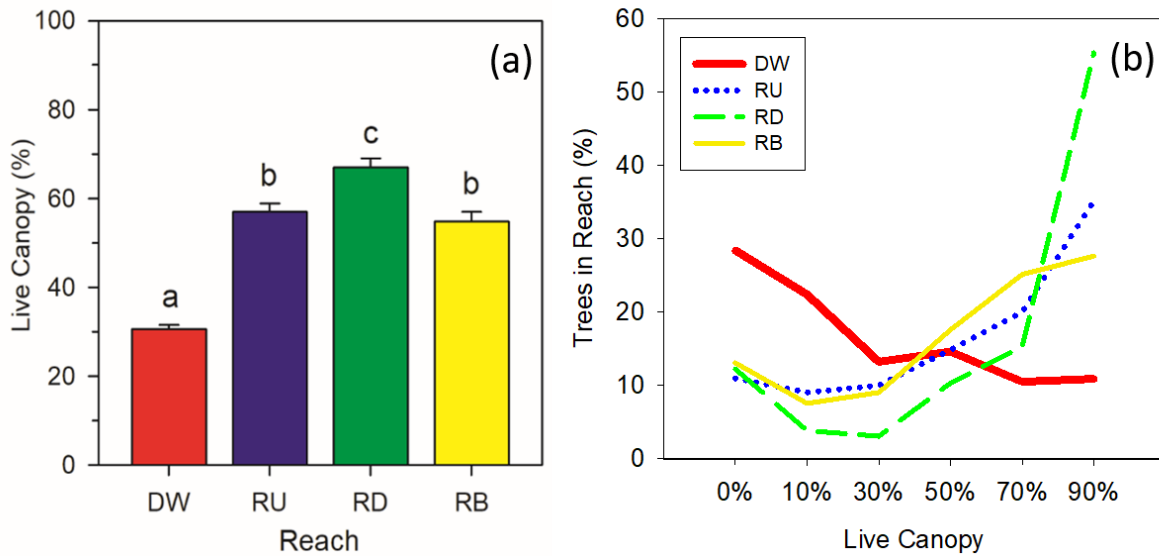


Figure 3.6. (a) Percent live canopy (mean + standard error) for all standing cottonwoods across reaches, with different letters showing significantly different means. (b) Proportion of trees in each of the six canopy classes per reach. In (b) most DW trees were in the lowest canopy classes (left), while reference reaches had mostly high canopy (right).

Investigation of age effects showed the youngest DW trees to have the highest mean percent live canopy at 80%, but the oldest trees (55%) did not have lower live canopy than middle-aged trees (51%; Figure 3.7). High canopy vigor for the youngest trees was expected because young trees generally grow rapidly. Further, their maturation and canopy expansion occurred mostly in the last half-century during stream dewatering. As such, they did not experience a change in hydrologic regime in the way that the old- and middle-aged trees did. Counter to expectations, the oldest trees did not have lower canopy vigor than middle-aged trees. This indicates that old age did not cause low percent live canopy. Furthermore, a group of very old cottonwoods in the DW reach not represented in this random sample dated back to 1782 so very old age is possible under the right growing conditions. Therefore, we do not believe age to be the cause of the low canopy vigor in the DW reach, and instead attribute the poor condition of the riparian forest in the DW reach to other environmental conditions.

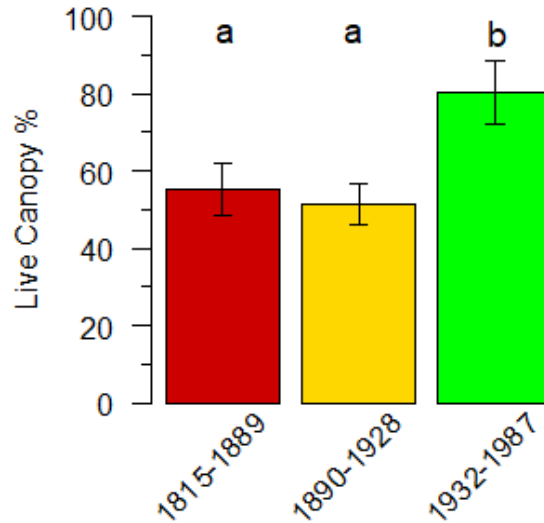


Figure 3.7. Percent live canopy among living trees in three age classes from the DW reach. Bars show mean live canopy with standard error around the mean.

The LAI measurements confirmed the visual assessments of canopy vigor. LAI values separated the six canopy classes into three statistically significant levels (Figure 3.8). Trees in the 10% canopy class had significantly lower LAI than trees in all higher canopy classes (30% to 90%). Trees in the 30% and 50% classes were significantly lower than trees in the 70% and 90% classes. The 90% canopy class had the highest average LAI. The mean LAI for each live-tree canopy class increased for every higher canopy class. These methods generally agreed with each other, and small discrepancies in the LAI classes likely occurred because the ceptometer was unable to differentiate PAR intercepted by live leaves, live branches, and dead branches. A linear model linking percent live canopy to LAI also showed a significant relationship between canopy class and LAI ($F = 57$, $p < 0.001$), also supporting our visual classification.

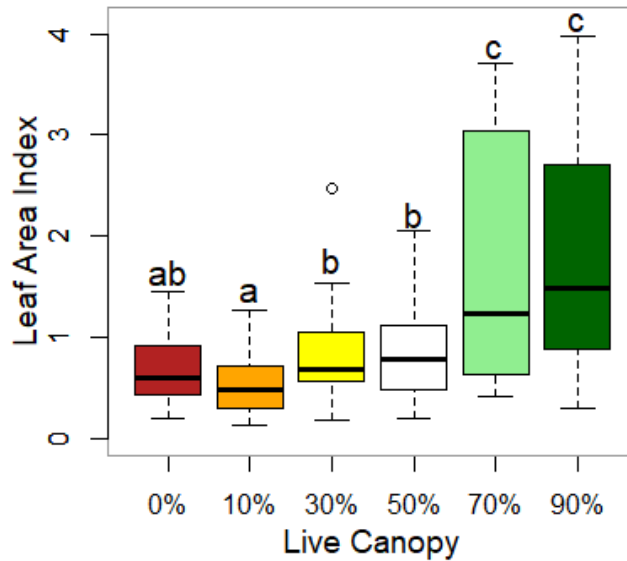


Figure 3.8. Leaf area index for the six canopy classes showing that the average value for each higher canopy class increases for all groups of living trees ($\geq 10\%$ canopy). Statistically different groups are shown by groups not sharing letters.

3.4 Cottonwood Dieback and Mortality

3.4.1 Patterns of tree mortality across reaches

3.4.1.1 Methods

Dead trees represented a substantial portion of identifiable cottonwoods in the DW reach, so we sampled them to characterize patterns and causes of tree mortality. In 2017 we randomly sampled 110 standing dead trees in DW to analyze growth patterns leading up to death. Because dead trees are difficult to sample with an increment borer, we extracted wedges at waist height using a chainsaw. The sample wedges were approximately 8 cm wide by 3 cm high and extended into the center of the tree if wood quality and aim allowed. All trees in this sample were free-standing and at least 2 m tall. Analyses were performed using standard dendrochronological techniques (Stokes & Smiley, 1968). Wedges were processed in the USGS Dendrochronology lab in Fort Collins, Colorado, where they were sanded, cross-dated, annual ring widths were measured, and cross-dating accuracy was validated using Cofecha software (Cook, 1985; Holmes, 1983). We assigned a tree's death date as that of the outer ring. Erosion of outer rings appeared to be absent or uncommon, although it is possible that the final growth ring was occasionally undetected. In 2018 we randomly sampled wedges from 22–26 standing dead trees at RU, RD, and RB using the same methods. Six wedges were not successfully cross-dated either because the wood was too rotten or because the ring width patterns did not conform to the reach's chronology.

In 2018, we sampled fallen dead trees at DW only by cutting wedges from 58 trees randomly selected from the 454 fallen trees identifiable as cottonwoods. Of these, a small proportion ($\sim 10\%$) were not sampled in the field because their wood was judged as too rotten to enable successful cross-dating. Six of the fallen samples were not successfully cross-dated, producing 52 fallen dead trees

with death dates. This yielded 229 standing and fallen riparian cottonwoods from which we determined the year of death.

We analyzed annual growth pattern of trees preceding their death using a modification of the regional growth curve concept (Briffa et al., 1992; 1996). We began by aligning each living tree ring series by tree age, with the first year from the pith being year 1 and counting outward. If a wedge missed the center, we followed Meko et al. (2015) to estimate the number of missing rings by dividing the radius of curvature of the innermost ring by the average width of the first four rings. We then corrected each ring-width value for tree age by dividing by the average value for trees of that age (Briffa et al., 1992, 1996; Meko et al., 2015; Schook et al., 2016b). Finally, we aligned age-adjusted ring widths by years before death instead of the usual years after establishment. We averaged data for all trees from a reach to find their average age-adjusted growth approaching death. Growth rates were expressed as a proportion of expected basal area increment (BAI), which characterizes cross-sectional tree growth and is described in Section 4.2.1. We compared DW standing, DW fallen, RU, RD, and RB trees. We evaluated statistical differences among reaches by grouping the annual values into decades for 1–10 years before death, 11–20, 21–30, etc. Cross-reach comparisons were made using the ten values from each decade and tested using analysis of variance with post-hoc Tukey's HSD testing.

3.4.1.2 Results and Discussion

Cross-dating revealed that cottonwood mortality at Snake Creek occurred between 1958 and 2017, and there was a slight tendency for trees near each other to have died at similar times (Figure 3.9). Most of the standing dead trees died from 2006–2017, while most trees dying earlier had fallen. The timing of mortality varied slightly by reach. Most standing dead trees in all reaches died in the last decade, but DW was the most extreme with 80% of mortality after 2005, compared to 58%, 58%, and 59% at RU, RD, and RB (Figure 3.10). Correspondingly, fewer standing trees at DW died earlier in time. For example, DW had 13% of trees die between 1990–2004, while RU, RD, and RB had 27%, 38%, and 35% of trees die during this period.

454 dead cottonwoods were identified to have fallen in the DW reach, which was 23% of all identifiable cottonwoods and 50% of all dead cottonwoods in the reach. The 52 fallen dead trees we cross-dated had death dates from 1958–2010, and dates were distributed much more evenly through time compared to those from standing dead trees. The largest mortality pulse occurred around 1970 (1967–1975), a decade after the pipeline was installed, and a second pulse occurred around 1990 (1986–1993) (Figure 3.10e). Few trees died in the early 1980's. This low-mortality period included 1983 and 1984, which had the two highest peak discharges in the 55-year record at nearby Cleve Creek (Figure 2.8) and were the two highest tree ring growth years for DW cottonwoods in the last 130 years. There was an unexpectedly well-defined threshold 15 years before sample collection when dead trees transitioned from free-standing to fallen. 84% of standing dead trees died in 2002 or later. Of fallen dead, however, 89% died in 2001 or earlier. Expanding the standing dead survey to include fallen trees greatly improved characterization of tree mortality patterns at DW and provided a better perspective on tree mortality. Combining the standing and fallen dead trees showed that there were three apparent pulses of death, which were centered around 1971, 1990, and 2009 (Figure 3.11).

Tree ring growth patterns revealed that fallen DW trees had significantly faster growth approaching death compared to dead standing DW trees (Figure 3.12a). For decadal comparisons of annual growth rates, the fallen trees had growth significantly closer to that of living trees for each of the last 5 decades preceding death (p 's < 0.05). Thus, the fallen trees died longer ago and more abruptly. Standing dead trees endured a slower decline preceding death. Standing dead trees had a median death date of 2008 and last achieved 100% of expected growth 49 years before death, in 1959. Fallen dead trees had a median death date of 1984 and last achieved 100% of expected growth 18 years before death, in 1966. These findings support the hypothesis that a change in water access in the 1960s affected tree growth leading to tree death, rapidly for some trees and more slowly for others.

Tree ring growth from dead trees at DW was significantly closer to growth from living trees compared to any reference reach in the decade before death (Figure 3.12b). In the last decade before death, DW dead trees grew at 53% of the rate for surviving trees of the same age. In contrast trees at RU, RD, and RB in the last decade before death grew at 26%, 39%, and 43% of the rate for surviving trees of the same age. Significant differences lasted for one decade at RB, two decades at RD, and seven decades at RU. The abrupt DW growth decline preceding death manifested as a steepening drop in annual growth in the 15 years preceding death (Figure 3.12b). The relatively abrupt death indicates that tree health declined faster at DW than at the reference reaches. The more prolonged decline at reference reaches is the reference pattern, representing patterns incited by multiple natural low-pressure environmental stresses contributing to death at different times.

If we evaluate growth preceding death at the three Snake Creek reaches only, DW's decline is even more clearly the most abrupt in the final decade before death (Figure 3.12). In contrast to DW, the final 25 years of tree life at RU and RD both had a concave up pattern, showing that trees remained alive for decades after the onset of decline. This pattern is more characteristic of what we would expect if old age or a slowly changing environment caused decline to death. However, the concave down pattern seen at DW signifies that mortality occurred relatively fast following the decline in growth, which is more characteristic of an abruptly altered growing environment.

At all sites, trees had successively smaller rings as they approached death compared to living trees from their reaches. Cottonwoods sacrifice branches when growing conditions become unfavorable (Tyree et al., 1994), which reduces leaf area and potential carbon assimilation that contributes to tree ring growth. Reduced ring width preceding death represents a tree's gradual decline. In contrast, abrupt death without a gradual decline would occur if a tree was struck by lightning, or if a water diversion completely and permanently separated the tree from its water source (Scott et al., 1999; Shafroth et al., 2000).

Cross-dating wedges from standing trees identified tree death up to 52 years before sampling, which extended back to 1965 and covers nearly the entire pipeline era. Cross-dating fallen dead trees identified death back to 1958. As cottonwoods age, decline, and die, their wood rots and eventually becomes too low quality to cross-date. The more time since death, the less likely a tree will yield a usable sample. Therefore, our sampling method has an inherent bias toward successfully cross-dating trees that died more recently. Although we do not know the proportion of trees that remained cross-

datable 1, 10, 20, 50, 100, etc. years after death, we did identify that some trees remained datable for over a half century which covers the entire pipeline period.

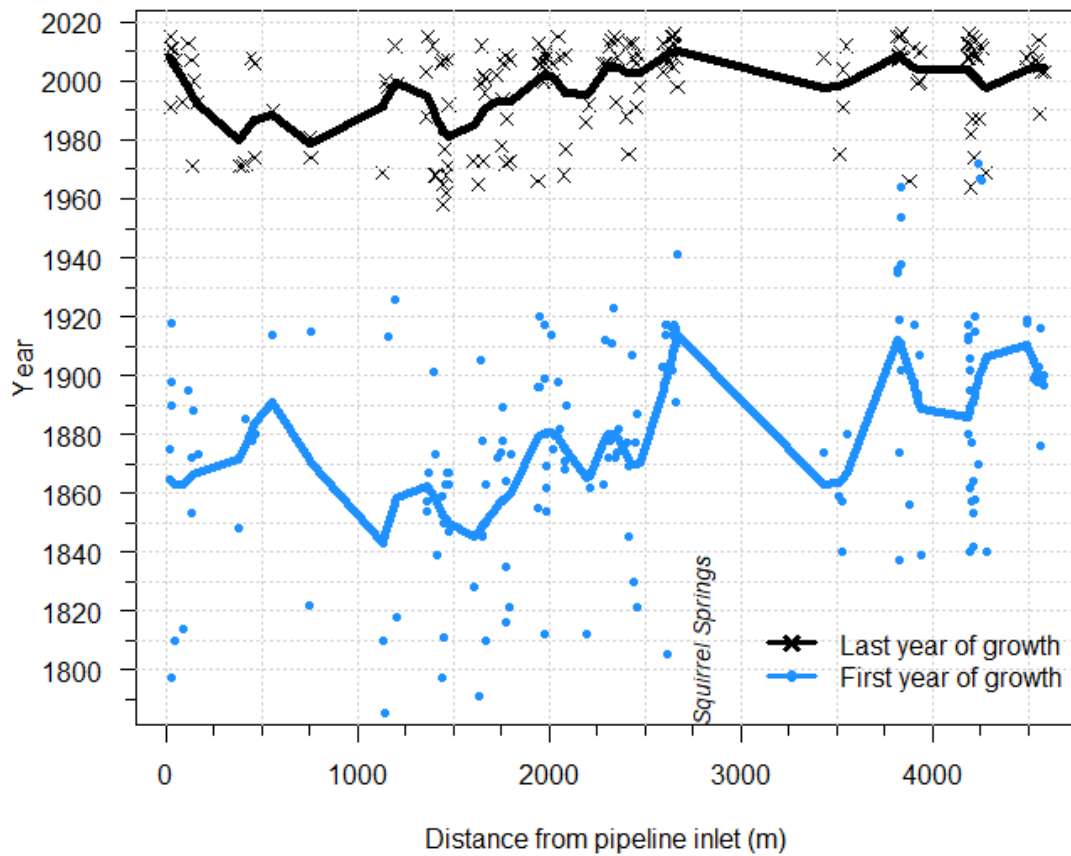


Figure 3.9. First and last year of growth for sampled standing and fallen dead trees in the DW reach, with lines showing smoothed averages by downstream distance. X-axis is distance (m) downstream of the pipeline inlet. Squirrel Springs is located at 2670 m and the pipeline outlet is at 4900 m.

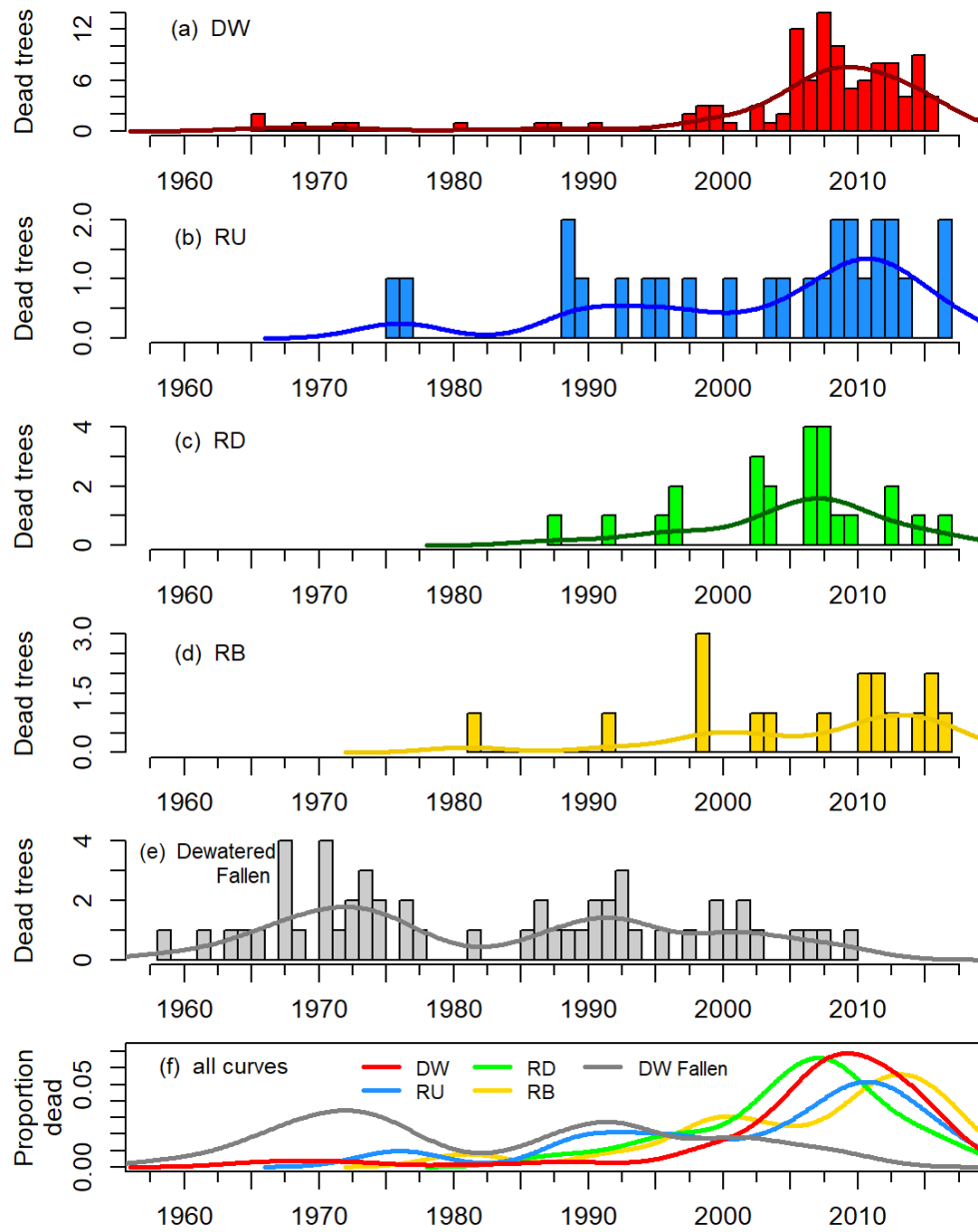


Figure 3.10. Death dates for standing dead trees in all four study reaches (a–d) and fallen dead trees in the dewatered reach (e). Panel F shows smoothed distributions standardized so that the area under the curve equals 1 for every group. Sample sizes are DW = 110 trees, RU = 26, RD = 24, RB = 17, dewatered fallen = 52. Note that y-axis scales differ.

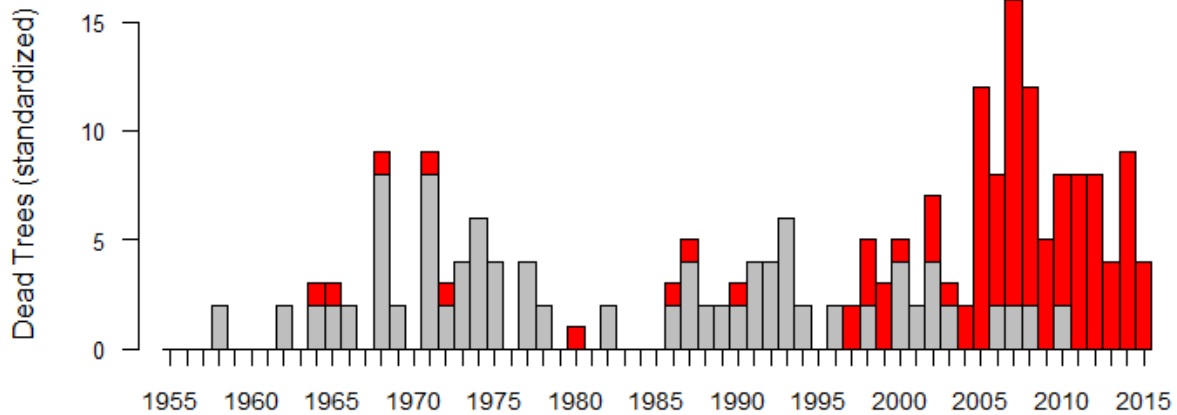


Figure 3.11. Death dates from all standing (red) and fallen (grey) trees in the Dewatered reach. Note that y-axis values were scaled to make the sum of each group equal because they were both found to represent 50% of the dead trees in the DW reach.

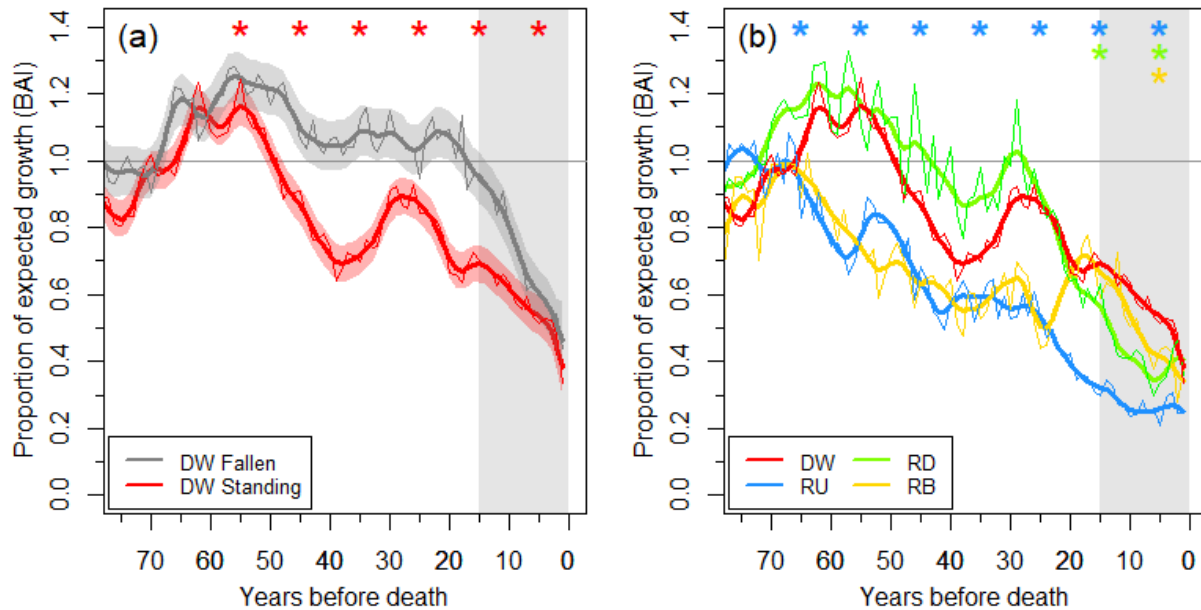


Figure 3.12. Tree ring growth as a function of years before tree death. Annual values (thin lines) are the mean of all trees in a reach, and smoothed lines are splines with a smoothing parameter of 0.35. Growth is shown as proportional basal area increment (BAI) compared to that expected from a tree ring of its age. (A) Dewatered reach fallen (gray) and standing (red) dead trees. (B) Standing dead trees from all four reaches. Colored asterisks show decades when DW Fallen (grey in A) or DW Standing (red in B) trees had higher growth compared to the other groups. Red = DW, blue = RU, green = RD, yellow = RB. In (B) note the concave down shape for DW trees and concave up shape for reference reaches in the final 15 years of growth (shaded).

3.4.2 Within-tree branch dieback

3.4.2.1 Methods

We felled and “dissected” 13 cottonwoods in the DW reach to reconstruct within-tree patterns of branch dieback and tree mortality. Eleven of these trees had 0% live canopy and two had < 5% live canopy, and we generally refer to these trees in this analysis as dead. Sampled trees were chosen to include the variety of morphological variation of dead standing trees in the DW reach. Before dissection, branch structure was evaluated and 9–18 locations on the tree were selected for dating. The number of dated locations varied by tree size and extent of branching. The dated locations were selected to encompass the variability of dead branch morphology on each tree.

Trees were felled and any broken branches were carefully replaced to match their orientation while standing. We then collected a wedge from the tree trunk and a disc from each branch for cross-dating ($n = 158$ samples). In the lab, trunks were cross-dated to the DW reach chronology to identify the tree death year. Each branch was cross-dated to the trunk, and the last growth ring was the branch death date. Each branch death date was compared to other branches from the tree to identify any anomalous dates, such as occurred five times from three trees when an upper branch death dated to after one of its supporting branches. This occurrence demonstrated that missing rings can occur if trees are stressed, but this is rare.

Death dates were compared to flow and climate variables to test the hypothesis that drought caused some of the branch death (Rood et al., 2000). We used the `regsubsets` function in R to create a multiple linear regression model using 16 flow and climate variables as possible predictors: average flow in May, June, March–July, and for the October–September water year; PRISM-derived data for annual precipitation, average maximum and minimum vapor pressure deficits, 5-year average maximum and minimum monthly temperatures, and 5-year average maximum and minimum monthly vapor pressure deficits (PRISM Climate Group, 2018); and Great Basin National Park weather station GHCND:USC00264514 for number of days over 21 and 32° C (70 and 90° F), annual precipitation, average temperature, and maximum temperature (NOAA, 2018). We explored all-subsets regressions using BIC and Mallow’s CP as the model selection criteria.

3.4.2.2 Results and Discussion

Death dates of the dissected trees were similar to those from the randomly selected DW trees (Figure 3.13, Table 3.3). Ten of the 13 trees (77%) lived until at least 2003 (Table 3.3), similar to 92 of 110 (84%) randomly selected trees. Likely because dissected trees were chosen to represent the variety of morphological structures of dead trees, the sample included a higher proportion of trees that died longer ago.

Branch dieback always preceded tree mortality, and death dates revealed a long history of tree dieback from 1903 to 2017. Branch death peaked in the 1960s and 1970s, roughly four decades before 2011, which was their median whole tree death date. High river flow and tree ring growth in the early 1980s coincided with reduced branch mortality, suggesting that when water is plentiful branches survive. Between the late 1980s and the 2017 sampling effort, there was relatively constant

branch dieback. In summary, branch dieback was much earlier and more dispersed in time compared to whole-tree death dates (Figure 3.14).

Branches from a given tree averaged dying 10–44 years (mean = 20 years, median = 14 years, $n = 145$) before the tree trunk, revealing a multi-decadal decline before death. Trees T02 and T10 declined relatively abruptly, with a standard deviation of 3 years among branch death dates. In contrast, T07 branches died from 1921–2003 and had a standard deviation of 24 years. These divergent patterns illustrate unique growing conditions, ecophysiological responses, and resilience within the population.

Linear regression analysis identified branch death dates to be inversely correlated to tree ring basal area increment ($F\text{-stat} = 4.3$, $p = 0.04$). In other words, years with wider tree rings had fewer branches die. Thus, better growing conditions supported radial expansion and prevented branch sacrifice, while poor growing conditions cause smaller tree rings and more branch dieback.

The multiple linear regression models linking branch death to climate did not reveal strong relationships. The Mallows' CP model ($R = 0.52$, $\text{Adj. } R^2 = 0.18$, $p = 0.03$) retained four variables but was judged to be overparameterized. The model retained 5-year averages of both maximum and minimum VPD, but their opposite loadings meant that they partially cancelled out. The other two retained variables were 5-year minimum temperature and number of days over 32 °C (90 °F), both of which were negatively correlated to branch death, counterintuitive relationships. The BIC model differed but was also deemed suspect ($R = 0.40$, $\text{Adj. } R^2 = 0.11$, $p = 0.05$). It retained two variables, March–July Q and days over 90 degrees F, both with negative loadings. Low spring runoff with high branch mortality is reasonable, but more hot days should not cause less branch death. Branch dieback in the DW reach in the 1960s and 1970s (Figure 3.14) is consistent with the hypothesis that atmospheric drought and flow diversion affected the forest. These weak models suggest, however, that branch death lags a year or more behind the effects of drought.

Trees are large and resilient organisms, and the within-tree branch dieback illustrated that whole tree death dates (Figure 3.11) do not capture all aspects of cottonwood decline. Many trees in the DW reach died in the decade following the onset of flow diversion in 1961 (Figure 3.11). Many others lost branches at that time, but survived for decades (Figure 3.14), eventually succumbing in later droughts (Figure 2.5, Figure 3.11, Figure 3.14). These results emphasize that tree mortality can occur gradually over decades in response to atmospheric drought and flow diversion.

Table 3.3. Timing of whole tree and branch death for the dissected trees. Tree death was identified as the trunk's last year of growth, and branch data are sample means for all branches.

Tree ID	Tree Death	Branch death (mean $\pm$ SD)	Tree death lag from mean branch death (yrs)
T01	2014	1990 $\pm$ 19	24
T02	2008	1994 $\pm$ 3	14
T03	1973	1963 $\pm$ 9	11
T04	2017	1998 $\pm$ 17	19
T05	2017	2005 $\pm$ 14	12
T06	2011	1967 $\pm$ 19	44
T07	2003	1973 $\pm$ 24	30
T08	1973	1960 $\pm$ 20	13
T09	2013	1995 $\pm$ 10	18
T10	1984	1972 $\pm$ 3	12
T11	2012	1985 $\pm$ 16	27
T12	2003	1985 $\pm$ 9	19
T13	2015	2005 $\pm$ 5	10
<i>Median</i>	<i>2011</i>	<i>1985 $\pm$ 14</i>	<i>18</i>

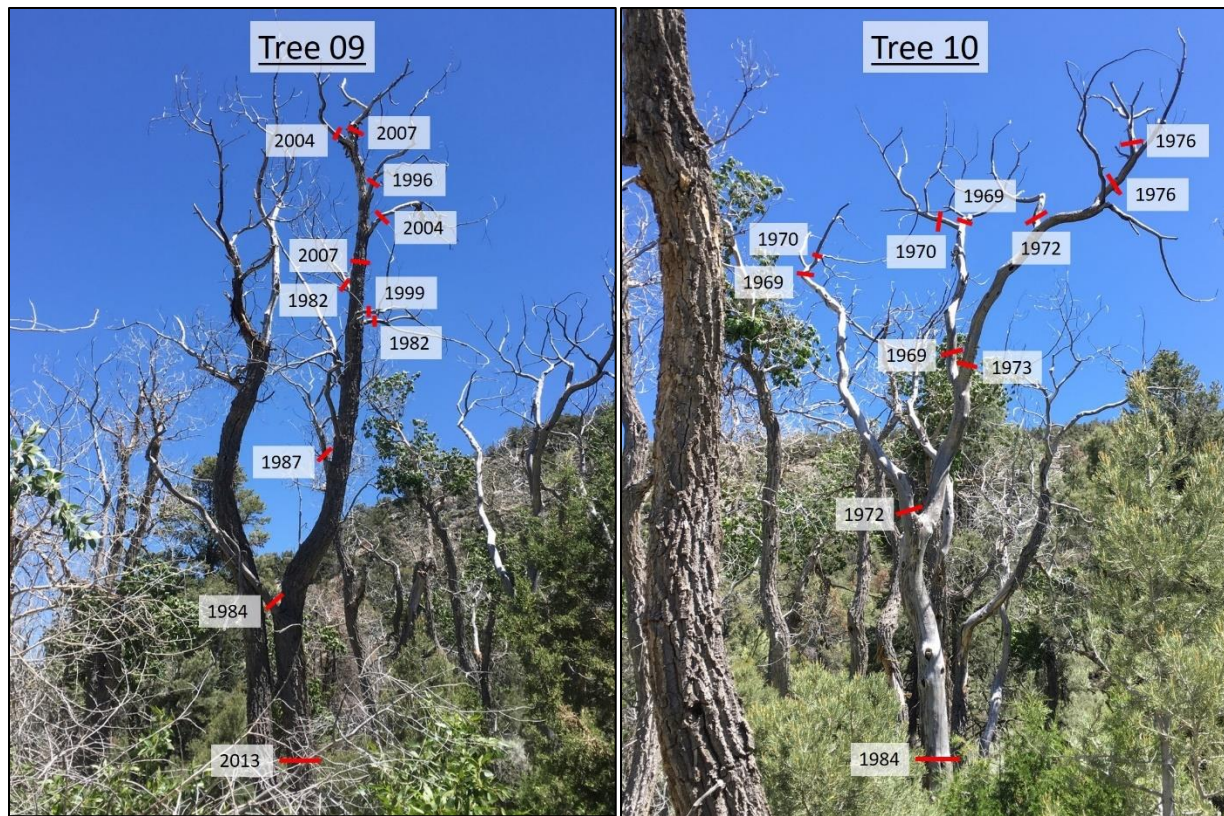


Figure 3.13. Photos of two example dissected trees showing the location of cross-dated branches and their death dates. Trunk death revealed tree mortality in 2013 (A) and 1984 (B), while the branches died between 1982 and 2007 (A), and 1969 and 1976 (B).

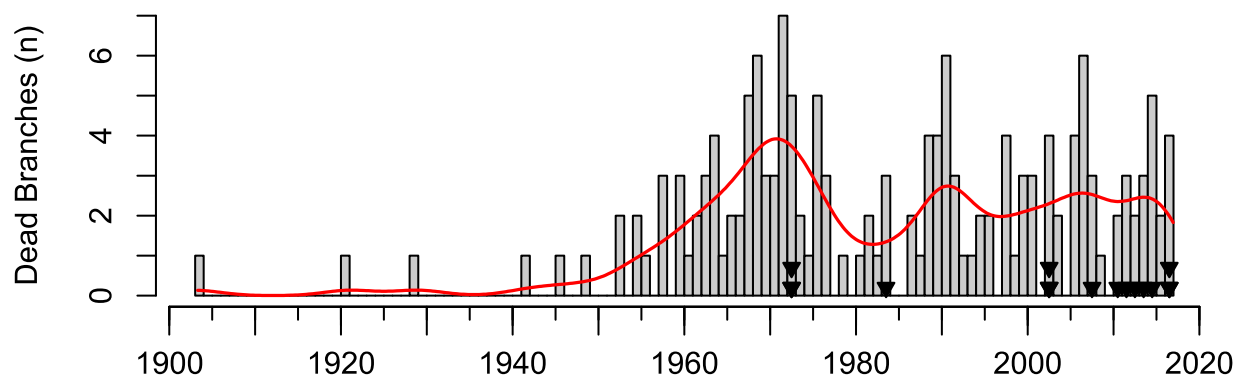


Figure 3.14. Death dates of the 158 sampled branches from 13 trees. Triangles show the last year of trunk growth for sampled trees. Red line is a kernel density function that smooths the histogram.

4. Tree Ring Chronologies

4.1 Introduction

Trees form annual growth rings whose characteristics vary based on growing conditions in the year of ring formation. Width is the most common tree ring metric analyzed because of the ease with which it is measured and because it is often highly responsive to changes in the local growing environment. The field of dendrochronology has identified various environmental factors that impact ring width, including temperature (Williams et al., 2013), pathogens (Andersen, 2015), fire (Schweingruber, 2012), and competition from nearby trees (Willms et al., 2006). However, in arid regions such as the Great Basin, annual variations in ring width are most commonly affected by growing season water access (St. George, 2014). Researchers have used ring width to identify both short-term (Schook et al., 2016b) and long-term droughts (Meko et al., 2007; Stahle et al., 2000) caused by factors such as climate (Gray et al., 2007) and water diversion (Schook et al., 2016a). By analyzing chronologies constructed from the annual growth patterns of cottonwoods in GRBA, we aimed to reconstruct historical growing conditions related to water access and identify any spatial and temporal variability across study reaches.

4.2 Methods

4.2.1 *Sample collection and processing*

Tree ring chronologies were developed from live cottonwoods for analysis of tree growth patterns through time. The 223 sampled trees were randomly selected from all living trees > 20 cm DBH in the DW reach and the three primary reference reaches (RU, RD, and RB). Random selection allowed the sample to represent average growing conditions in each reach. We developed a fourth reference reach called Reference Squirrel (RS), which was located in a 150 m subreach of the DW reach below Squirrel Springs where most trees had high percentages of live canopy and riparian vegetation was uncommonly dense. The RS chronology was constructed from nine randomly selected and eight easily accessible trees. The RB reach was restricted to the 15 trees from the lower sample area because it was post hoc identified as a better reference for the DW reach compared to the upper sample area that had widespread hillslope-derived groundwater. For DW we used a double-stratified sampling approach. Twelve trees were randomly selected from the upper and lower halves of the reach for each live tree canopy class (10%, 30%, 50%, 70%, and 90%). All of the DW cores were used in secondary analyses comparing canopy classes and subreaches. However, the main DW chronology was created from a randomly selected subset of these cores that were retained to match the proportional abundance of each canopy class. This reduced the 221 cores from 118 trees to 114 cores from 78 trees. Thus, the DW, RU, RD, and RB chronologies fully represented living trees in each reach, while RS was created from the more common dendrochronological approach of opportunistic sampling.

Tree ring chronologies were developed using standard techniques (Stokes & Smiley, 1968). Two cores were collected from each tree 1.3 m above the ground using increment borers with 5.15 or 12 mm inside diameters. The larger borer was used to collect samples for isotope analysis (Section 5). Cores were mounted and sanded using progressively finer sandpaper between 120 and 1200 grit (Figure 3.1). Cores were cross-dated visually under a dissecting microscope using skeleton plots,

measured using a Velmex Manual UniSlide micrometer and MeasureJ2X software, and quality controlled using COFECHA (Grissino-Mayer, 2001; Holmes, 1983).

We combined ring width and tree diameter to estimate basal area increment (BAI), the area of new wood added to the stem cross-section in each year (Biondi, 1999). BAI can be a more accurate reflection of whole tree growth than ring width and can reduce the need for detrending to remove the effect of tree age on growth (Biondi & Qeadan, 2008). If a core did not contain pith (i.e., tree center), we estimated tree diameter using methods from Meko et al. (2015). Sampled trees had a median of four missing rings, ranging from 0 for cores with pith to 95 for a rotten tree. BAI increased steadily during the first 25 years of growth but was relatively stable afterwards. Therefore, we retained only rings from when trees were at least 26 years old to remove the effect of tree age from our analysis. While ring width generally decreases as a tree ages and its radius expands, BAI usually remains similar or increases while trees are in good condition (Biondi & Qeadan, 2008). Therefore, a declining BAI is particularly diagnostic of poor tree condition and indicates increased risk of mortality (Bigler & Bugmann, 2003, 2004; Wyckoff & Clark, 2000).

4.2.2 Tree ring chronologies

Tree ring chronologies were used to compare different groups of trees. The primary comparison was for BAI among DW and the four reference reaches. We also standardized BAI to present tree ring growth in a way that facilitated comparisons to DW. In this standardization, all BAI's were first divided by the BAI of the DW reach in each year to calculate proportional growth. Each reach's proportional chronology was then shifted to have its 1901–1961 baseline period average 1. We shifted chronologies to 1 because it was a more conservative adjustment than dividing them to reach the same baseline value of 1. This procedure made the pre-pipeline period both standardized and relative to DW, enabling post-pipeline BAI to reflect changes from baseline compared to DW. All tree ring chronologies were created using the `bai.in` and `chron` functions in the `dplR` library in R (Bunn, 2008). Chronologies were calculated from raw BAI series instead of the common method of detrending because detrending removes low-frequency signals that were of interest (Briffa & Melvin, 2011; Schook et al., 2016b). Series were combined using biweight robust means, and chronologies are presented as smoothed splines of annual BAI to improve visualization and to facilitate comparison of multi-year trends.

Analysis of variance was used to test if BAI differed among sites in each period. T-tests were used to test if within-reach BAI changed between 1933–1961 and 1962–2016. The shorter baseline period was selected to restrict quantitative analysis to those years where all chronologies had ≥ 12 cores.

We also analyzed BAI chronologies that were constructed from four additional categories of DW cottonwoods. First, living and dead DW trees were compared to see if tree ring signals could help explain the death of some trees and survival of others. Second, percent live canopy classes were compared to evaluate if canopy vigor and basal production corresponded. Third, tree age groups were compared to investigate if dieback was related to old age, complementing the age group canopy analysis (Section 3.2.2). In the age group analysis trees were separated into three groups with establishment years 1815–1889, 1890–1928, and 1932–1987, and we compared BAI for 1930–2016, when all groups had ≥ 7 cores. Finally, five subreaches within the 5 km DW reach were compared to

identify if BAI production differed within the reach. Subreaches were ordered from 1 to 5 upstream to downstream and had 24–83 cores each (Figure 4.2).

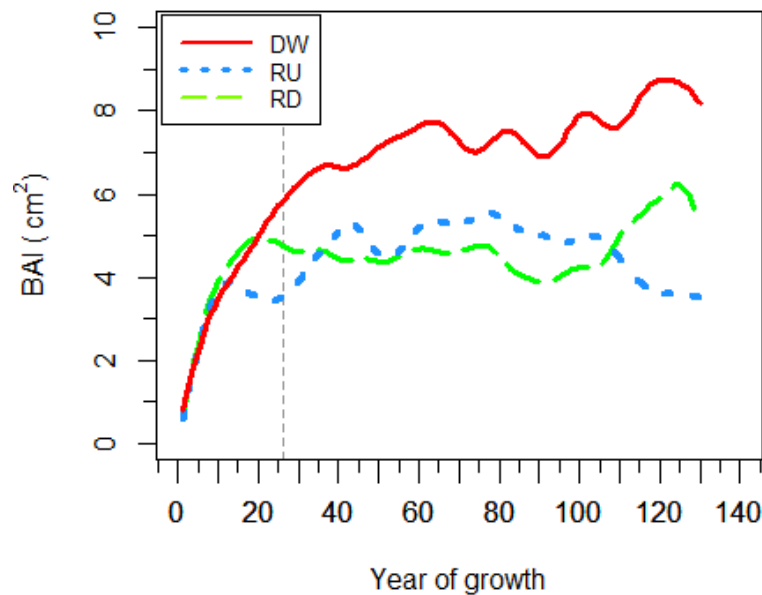


Figure 4.1. Basal area increment (BAI) as a function of tree age for DW, RU, and RD. There was a relatively rapid biological increase when trees are young. Dashed grey line at 26 years old is where curves stabilize; all chronologies only retained rings ≥ 26 years.

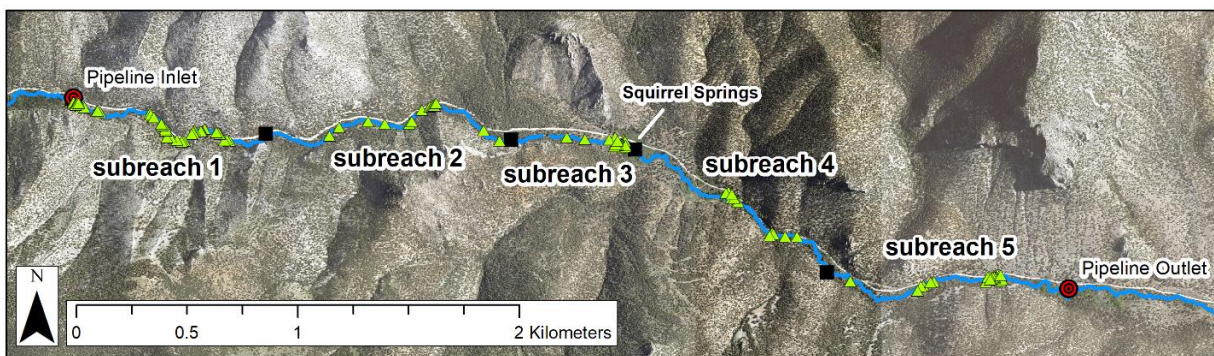


Figure 4.2. The five cottonwood subreaches in the dewatered reach. Subreaches are separated by black squares and the green triangles are cored trees.

4.3 Results

4.3.1 Descriptive statistics

All five reaches had at least one tree core with measured rings back to 1886 (Table 4.1). The average tree core series length was 77–106 years across reaches, which was less than the average tree age because each series included only rings ≥ 26 years old. Interseries correlations ranged from 0.48 to 0.64. These correlations combined with large sample sizes yielded high confidence in the

environmental signals contained in the chronologies, which was quantified by the expressed population signals (EPS) ranging from averages of 0.86 to 0.96 across reaches.

Table 4.1. Dendrochronology descriptive statistics for tree ring chronologies from each reach. Note that the DW reach chronology was for proportional subsampling.

Measurement	Chronology				
	DW	RU	RD	RS	RB
Trees (n)	78	24	28	17	15
Cores (n)	114	46	53	34	29
Year range	1844–2016	1845–2016	1867–2016	1886–2016	1868–2016
Mean series length (yr) ¹	88	106	77	98	80
Interseries correlation	0.51	0.58	0.48	0.61	0.52
AR1 ²	0.63	0.56	0.64	0.66	0.65
Mean BAI (cm ²) for 1900–2016	7.42	4.98	5.03	6.20	5.58

¹A tree core series included only rings that were at least 26 years old. ²AR1 is first order BAI autocorrelation.

4.3.2 The dewatered (DW) reach chronology

The DW chronology revealed several periods of high and low tree ring growth preceding a sharp decline after the early 2000s (Figure 4.3). EPS exceeded 0.83 for the entire period, indicating high confidence in the chronology. BAI averaged 7.42 cm²/yr from 1900 to 2016, and it ranged between an annual low of 2.41 cm² in 2015 and a high of 11.86 cm² in 1983.

BAI averaged 8.55 cm²/yr for 1916–1961, which was selected as the baseline growth period because it was within 45 years of pipeline installation and EPS remained > 0.85 throughout this period (Wigley 1984). A decline occurred around 1960, and from 1962–1979 BAI was only 77% of baseline (Figure 4.3). From 1980–1988, a period of high precipitation and flow, BAI increased to 108% of baseline. Every year after 1989 was below the average growth of the baseline period. BAI averaged 73%, 73%, and 46% of baseline in the 1990s, 2000s and 2010s. The four lowest-growth years from 1916 to 2016 were the last four (2013–2016), when BAI averaged 2.89 cm²/yr (34%) of baseline. The two years with highest growth were 1983 and 1984 (Figure 4.3), which coincided with the two highest peak discharges at Cleve Creek (Figure 2.8). The third highest peak discharge was in 2005, which had the highest post-1988 BAI for Snake Creek cottonwoods. However, the 2005 BAI was still below average compared with the baseline period. Similarly, 2011 was the fourth highest flow year at Cleve Creek (Figure 2.8) and the tree ring chronology again had a peak in BAI that was even farther below the baseline average. These low BAI peaks in the 2000s partly result from flow diversions that begin in 1961 in Snake Creek, but not Cleve Creek. The progressive decline in BAI in the 2000s indicates a recent limited ability of trees to respond to wetter years, similar to findings in *Populus fremontii* on the Green River (Williams & Cooper, 2005). It appears that dieback and sustained stress have lowered tree ring growth potential.

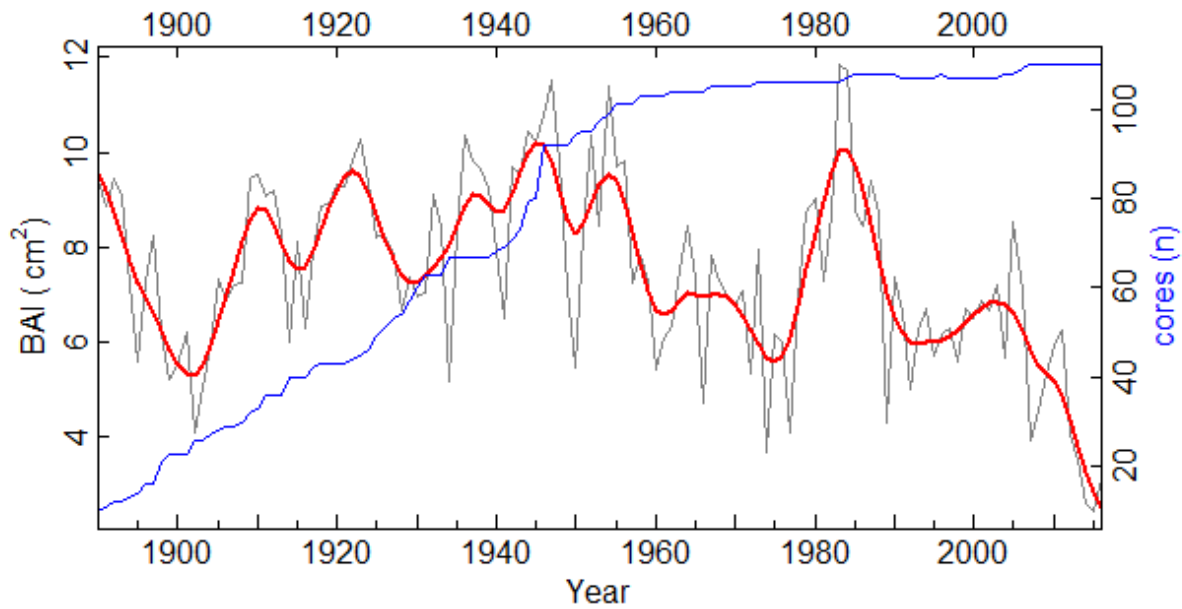


Figure 4.3. Tree ring chronology for the Dewatered reach, 1890–2016. Gray line is annual chronology, red line is 10-year smoothing spline, and the blue line on the right axis is tree core sample size.

4.3.3 DW and reference reach chronologies

Annual BAI production varied among the five study reaches in the pre- and post-pipeline periods (ANOVA: $F = 46.4$ and 39.5 , p 's < 0.001 ; Table 4.2). In the 1933–1961 baseline growth period, selected because all chronologies had ≥ 12 cores, the DW reach annual BAI was 1.87 times the average BAI of the reference reaches (8.68 vs. 4.64 cm^2/yr). In the 1962–2016 post-pipeline period, however, the DW growth rate was only 1.14 times greater than the reference reach BAI's (6.53 vs. 5.73 cm^2/yr). Thus, DW had much higher cottonwood growth in the pre-pipeline period but only minimally so in the post-pipeline period. The BAI declines between pre- and post-pipeline periods were similar for DW and RU with reductions to 75% and 72% of baseline growth, respectively. RD had a small increase to 108% of baseline growth in the post-pipeline period. RB and RS were similar to each other in both periods but different from the other three reaches, increasing to 144% of baseline for RS and 202% for RB in the post-pipeline period. The high variability among reference reaches (Table 4.2) illustrated that interacting environmental factors beyond flow regime influence tree growth.

Table 4.2. Average BAI for the study reaches for 1933–1961 and 1962–2016. Significant differences between periods within a reach are denoted by “*” at $p < 0.1$ and “**” at $p < 0.001$.

Reach	Pre BAI (cm ²) (1933–1961)	Post BAI (cm ²) (1962–2016)	Pre-Post change BAI (cm ²)	Post:Pre BAI ratio
DW	8.68	6.53	-2.15**	0.75
RU	5.86	4.17	-1.69**	0.72
RD	4.11	4.43	+0.32*	1.08
RB	3.34	6.75	+3.41**	2.02
RS	5.26	7.57	+2.31**	1.44
Reference Average	4.64	5.73	+1.09**	1.23

Comparing tree ring chronologies through time showed that cottonwood growth differed by reach. The DW reach had the highest BAI every year from 1930 to 1961 (Figure 4.4a). After 1962, however, DW had the highest BAI in only 8 of 51 years. RB or RS were the highest in the other 43 years. In contrast, RU and RD maintained their relatively low baseline growth rates during the post-pipeline period. BAI in the dense cottonwood forest at RU only surpassed the BAI DW in 1975, 1977, and 2015, and BAI at RD only surpassed DW in 1995, 2007, and 2012–2016. Because most of these years were isolated, they do not appear higher than DW in the smoothed trends in Figure 4.4a. In summary, raw BAI comparisons indicated that RB and RS were greater than DW in the post-pipeline period but RU and RD were not.

Standardization of reach-level BAI showed each reference reach to have higher growth compared to DW after 1961 (Figure 4.4b; Appendix Figure 16.1). The patterns emerged in the 1960s for RB and RS and in the late 2000s for RU and RD. RB and RS had growth averaging 1.55 and 1.61 times higher than DW in the entire post-pipeline period. In contrast, RU and RD had nearly identical standardized growth to DW after 1961 at 0.98 and 1.03 times. The RU and RD chronologies didn’t diverge from DW until around 2007, which was caused by the DW growth decline (Figure 4.4a). After 2007 all reference reaches had higher growth compared to DW.

RS was a unique reference reach because it is located *within* the DW reach. Thus, the growing environment of RS is completely surrounded by that of DW, and tree ring growth patterns would be expected to resemble each other. However, field appearance and tree ring analysis indicate that RS trees accessed a water source other than Snake Creek surface streamflow. In the pre-1961 period, the DW and RS chronologies closely resembled each other, as high and low growth periods coincided (Figure 4.4a). However, this relationship was decoupled after 1961. Most notably in the 1960s and 1970s RS growth increased substantially, while it decreased at DW. In the mid-1990s RS growth declined rapidly before rebounding in the mid-2000s. Both of these swings were muted for DW trees. Although the higher variability at RS could relate to sample size, the discrepancy in the two decades after pipeline installation is likely due to water access.

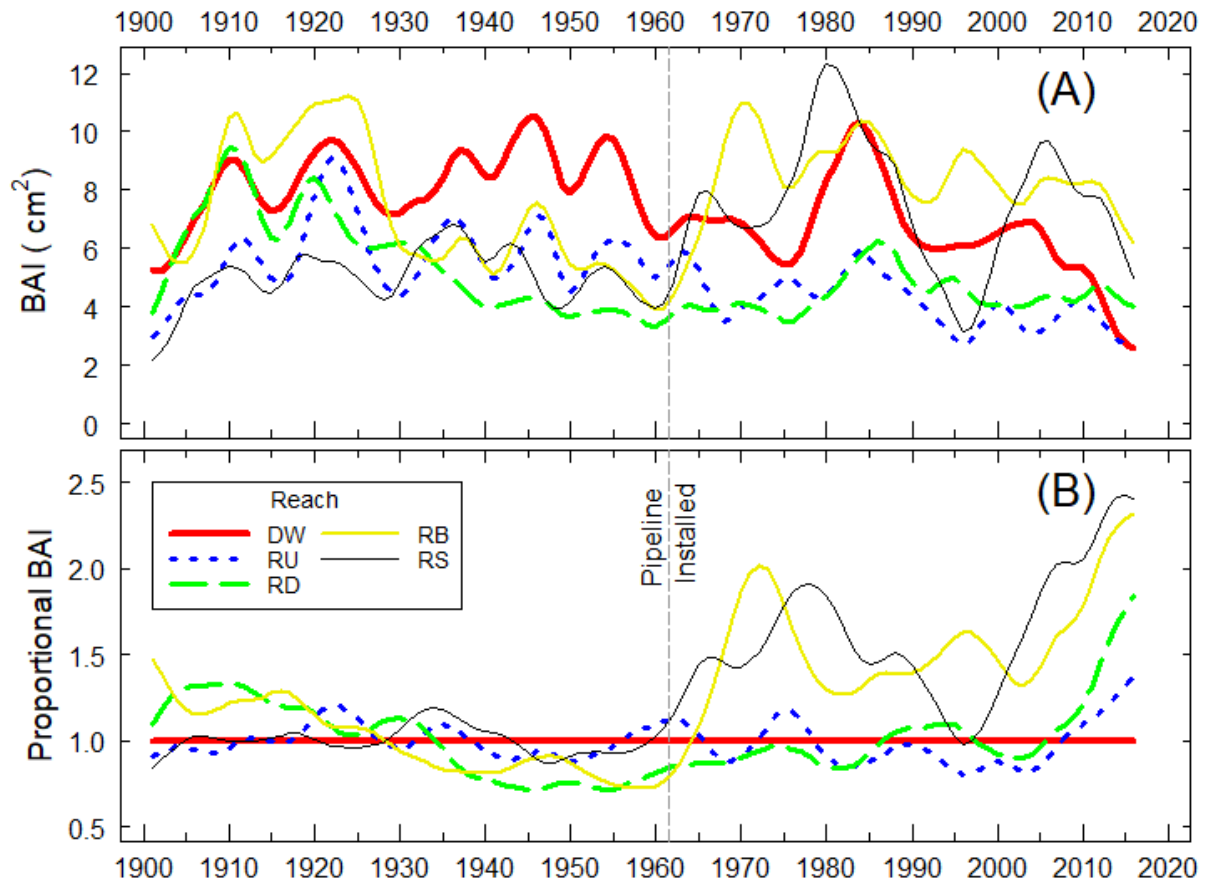


Figure 4.4. (A) smoothed tree ring chronologies from the DW and four reference reaches from 1901 to 2016. (B) is the same information scaled to show BAI relative to that of DW, and each reach's BAI is standardized to its baseline growth rate. Note that absolute BAI was highest at DW for most of the baseline period (A). At RB and RS relative growth surpassed DW around 1961 while a similar shift occurred at RU and RD around 2007 (B).

4.3.3.1 Discussion on reach differentiation

BAI differences across reference reaches highlighted the natural variability in GRBA cottonwood growth. Water access is one of many factors affecting tree ring growth, and reach-level dynamics affect annual and multi-year growth rates too (Stromberg and Patten 1992). For example, the low growth period at RB in the 1930s to 1950s could have occurred due to cottonwoods shading each other, resulting in the death of some trees and a competitive release in survivors that led to BAI increase identified in the 1960s. At DW the branch mortality and dead-tree analyses showed high dieback in the 1970s. Decreased water access that caused the dieback of some trees and led to the competitive release of survivors likely caused the 1983–1984 BAI spike to be higher at DW than any reference reach (Figure 4.3 and Figure 4.4).

The RU chronology was unexpectedly similar to the DW chronology, and factors unrelated to stream dewatering must have influenced its low tree ring growth. Cottonwoods in the 5-km DW reach occupied a variety of microhabitats. RU trees, however, nearly all grew in a dense forest with high

competition, and they were from a relatively even-aged cohort dating to 1876–1899 (Figure 3.4). High tree density limits light access and photosynthesis, and the cooler temperatures found in the high elevation RU slow biological processes. Anecdotally, the RU and RD forests were the densest of the five reaches, which could have also affected their growth rates through time. The RB and RS forests had trees more similarly spaced as DW trees, these two reference chronologies had higher tree ring growth than DW beginning early in the pipeline period.

4.3.4 Tree ring growth by canopy class

The chronologies for each DW canopy class revealed that the percent live canopy encountered in field surveys related to BAI over most of the last century (Figure 4.5). The smoothed chronologies were perfectly ordered after 2000, as BAI was highest for the 90% canopy class and successively lower for each decreasing canopy class of 70%, 50%, 30%, 10%, and 0%. Trees dead in the 2016–2018 field surveys (0% canopy) had the slowest growth for most of the last century (Figure 4.5 and Figure 4.6), which indicated that factors such as marginal microhabitat and long-term stress disadvantaged the trees long before their death.

The lowest and highest percent canopy classes, 0% and 90%, had the lowest BAI declines following the high growth 1980s (Figure 4.5), but the declines occurred in different ways. The 0% trees had the lowest growth in the 1980s, and therefore the lowest potential BAI reduction. In contrast, the 90% trees were most able to maintain their 1980s high growth through the subsequent decades. The 90% canopy class had the most unique patterns through time among the canopy classes, having the largest BAI increase from 1962 to 1964, the smallest decline around 1975, and the smallest decline from 1985 to 2016 (Figure 4.5). The recent relative tree ring productivity of these 90% trees contrasts with the pre-pipeline period when their growth was intermediate among the canopy classes. This transition from average to high growth suggests that in the pre-pipeline period, when all trees had natural water access, the microhabitats of the 90% trees were not particularly advantageous. However, in the period of stream dewatering, these trees emerged as the most productive at both the canopy and tree ring levels. This pattern suggests that many of the 90% trees access a water source other than groundwater recharged by flows, which has elevated them to be most capable of surviving in the current flow regime.

BAI differences among the six canopy classes amplified beginning in the high-growth 1980s. The standard deviation among the classes quantified how different ring widths was by class, and it increased substantially from 1.41 cm²/yr in the 1970s to 2.36 cm²/yr in the 1980s (Figure 4.7). Unlike the BAI convergence that led to low standard deviation during the low-growth 1960s and 1970s, canopy class separation remained high in the low-growth 1990s–2010s. This suggests that long-term differentiation intensified in the 1980s, possibly because limb dieback and tree mortality had already begun.

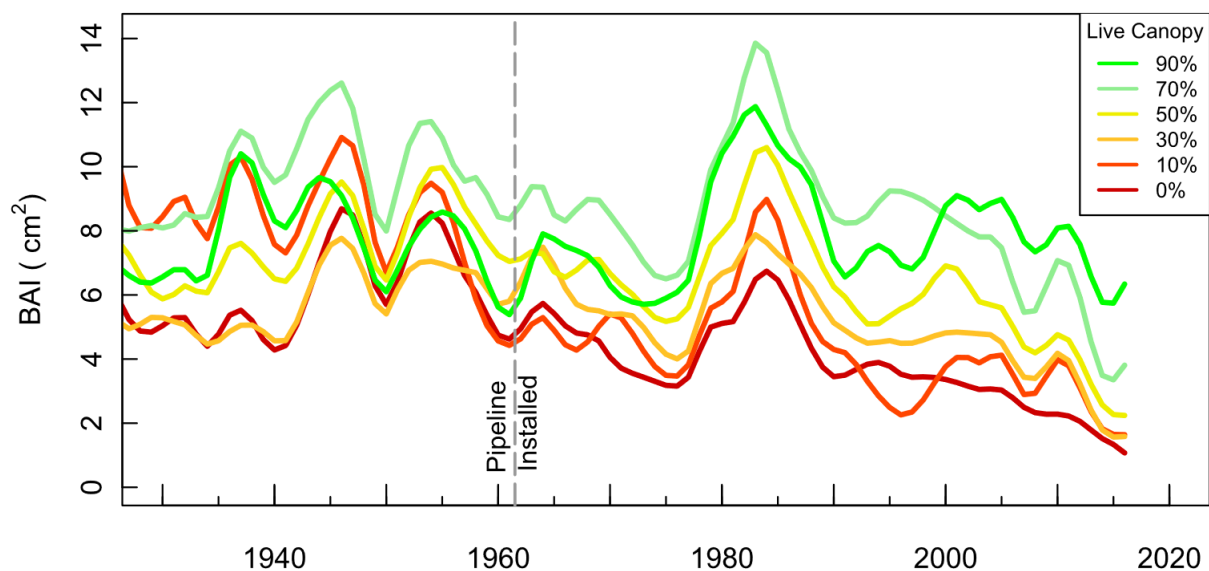


Figure 4.5. Tree ring chronologies for the six canopy classes in the DW reach. Annual BAI values are splined using a smoothing parameter of 0.3.

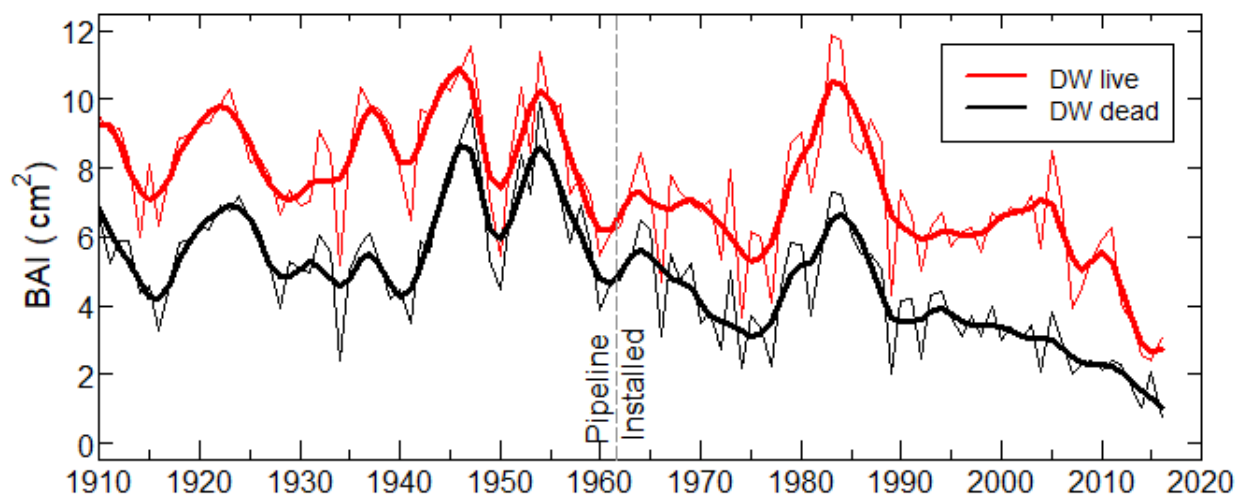


Figure 4.6. Dewatered (DW) reach chronologies constructed from living trees (red) standing dead trees (black). Thin lines are annual values and thick lines are splined averages using a smoothing parameter of 0.3.

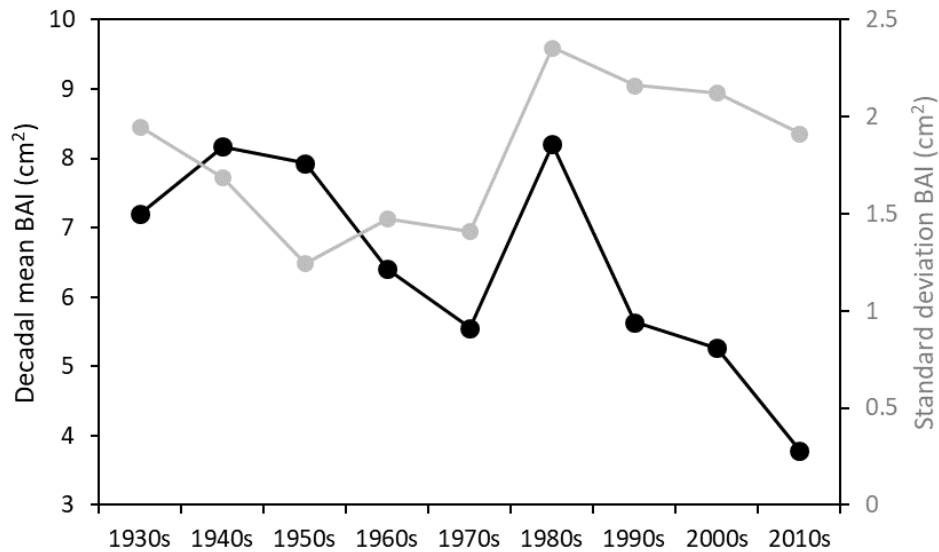


Figure 4.7. Decadal mean BAI (black) for DW trees and the standard deviation (gray) among the six canopy classes. Standard deviation results indicated that separation among canopy classes has been large since the high growth 1980s.

4.3.5 Tree ring growth by age group

Separating living DW trees into three age groups revealed small yet notable differences (Figure 4.8). Although age groups had similar BAI for most of the pipeline period, the youngest trees had the highest growth in the 1990s–2010s. This mirrored their high percentage of live canopy (Figure 3.7). This higher growth in young trees was not a function of their age because BAI generally increased with tree age (Figure 4.1). Instead, it is more likely that because the young trees matured under dewatered conditions, their roots and branches developed in accordance with the reduced water. In contrast, root and branch structure in the older and middle-aged trees were already established when the flow regime changed, which likely required adjustments that reduced tree vigor and BAI.

The oldest trees had higher growth than middle-aged trees for 98 of the last 100 years. Old trees would have had the lowest BAI if factors associated with old age caused the recent decline. Because this was not the case, we conclude that the documented BAI and canopy declines at DW were not driven by old age, which supports our conclusions in the canopy condition by tree age analysis (Section 3.3.2).

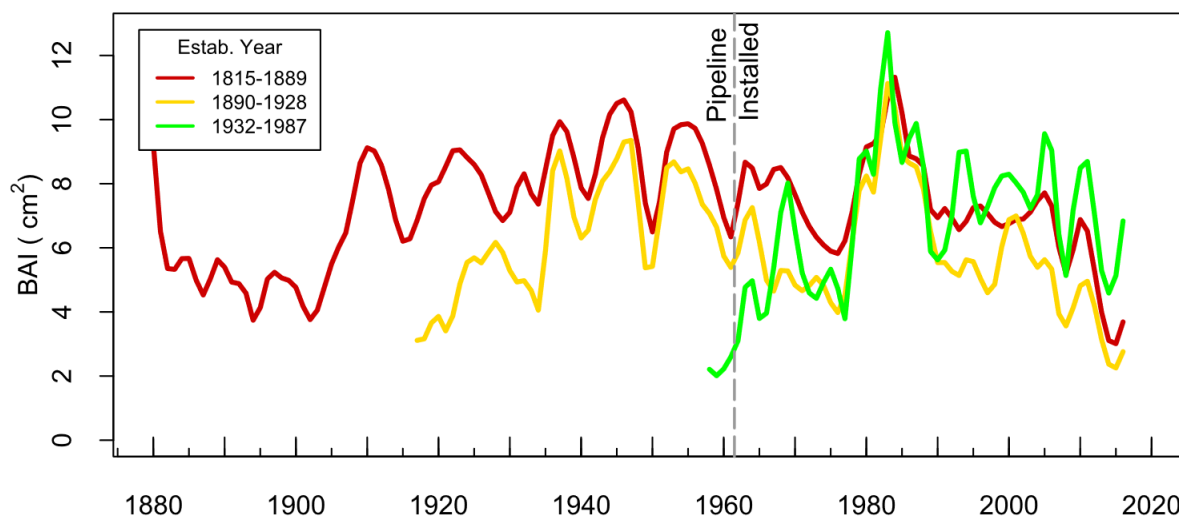


Figure 4.8. BAI through time for three age classes of Dewatered reach cottonwoods, with the annual values displayed as smoothed splines for visualization.

4.3.6 Tree ring growth in DW subreaches

There was no downstream trend in BAI among the five DW subreaches. Subreach 1 was the most upstream and had the highest BAI averaging 10.29 cm²/yr for 1930–2016. Subreach 3 was in the middle of the reach just above Squirrel Springs and had the lowest BAI at 4.56 cm²/yr (Table 4.3). Subreaches 2, 4, and 5 had similar BAI's that were intermediate, ranging from 5.94 to 6.79 cm²/yr. BAI at subreaches 3 and 5 had large declines of 33% and 41% between the pre- and post-pipeline periods, 1930–1961 and 1962–2016. In contrast subreaches 1, 2, and 4 changed less than 10% between the periods. Multi-decade BAI declines also varied by subreach between the high growth 1980–1988 and low growth in 2007–2016. Between these extreme periods subreaches 2, 3, and 5 all had large BAI reductions of 58–61%. Subreach 4, which included the RS reference reach, had the lowest decline at 34%. This relatively small decline supports the conclusion that the local groundwater discharge detected in field observations and indicated in electrical resistivity surveys (Section 10) provided an alternative water source in this area.

Chronologies were created for each subreach, which revealed additional differences among them (Figure 4.9). For example, subreach 2 was the only one to have a large BAI increase around 1945, and a similar increase was seen only at subreach 4 around 2000. These BAI variations unique to a subreach illustrated that the Snake Creek DW reach harbors a diverse and complex growing environment.

Variability in subreach BAI highlighted that spatial heterogeneity occurs on scales smaller than the DW reach. The reach is adjacent to the pipeline, but it is likely that its trees have been unequally affected by pipeline operations. For example, snowmelt runoff often overtops the pipeline inlet and flows into DW at subreach 1. If the runoff is not high and sustained, the surface water infiltrates into the alluvium before reaching downstream subreaches. Additionally, groundwater discharge points supply water to isolated trees in the DW reach (also see electrical resistivity, Section 10). Some of

these inputs occurred in subreach 4, where springs supplied an alternative water source to the apparently healthy trees.

Table 4.3. Mean BAI (cm^2) from the five DW subreaches for select periods before and after 1961. Values were calculated from all live trees cored in each subreach. The last three rows characterize the very high growth in the 1980s and low growth in the 2000s.

Period	DW Subreach				
	1	2	3	4	5
1930–2016	10.29	6.67	4.56	6.79	5.94
1930–1961	9.97	7.04	6.13	6.41	7.53
1962–2016	10.47	6.46	3.64	7.01	5.02
1930–61 to 1962–2016 difference	+5%	–8%	–41%	+9%	–33%
1980–88	13.06	9.32	5.81	10.35	8.18
2007–16	7.64	3.67	2.31	6.82	3.40
1980–88 to 2007–16 difference	–41%	–61%	–60%	–34%	–58%
Cores (n)	83	24	37	27	50

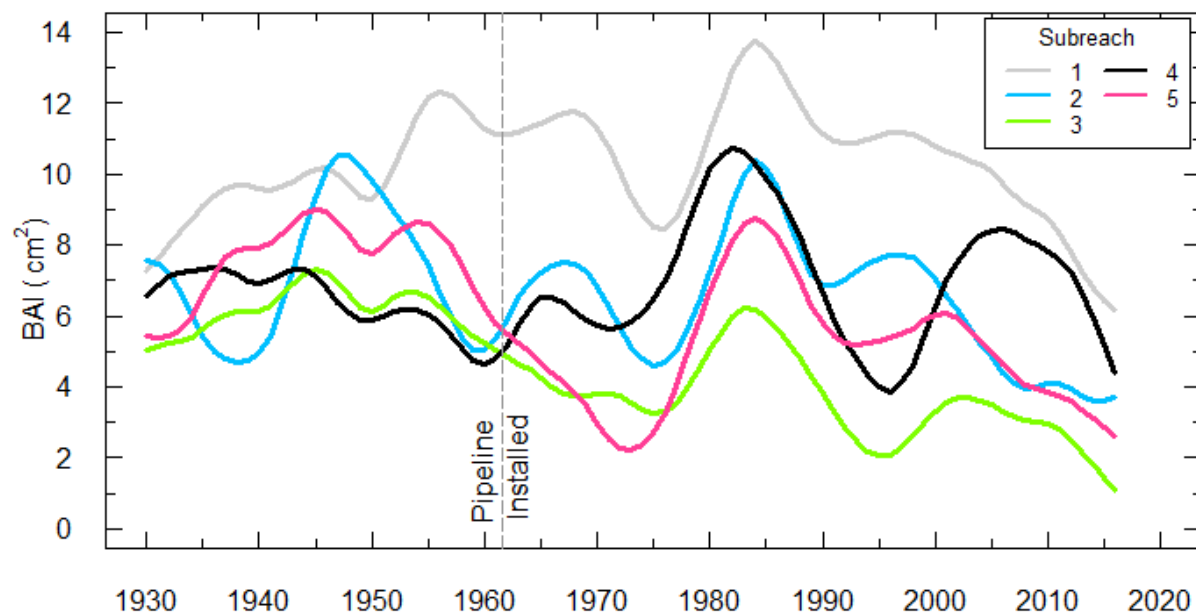


Figure 4.9. Tree ring chronologies for the five DW subreaches, from subreach 1 (upstream) to 5 (downstream). Annual BAI values were splined using a smoothing parameter of 0.5 for visualization.

4.4 Synthesis of Tree Ring Chronologies

Our comparisons among various groups of cottonwoods revealed high spatial and temporal variability in tree ring growth. DW tree ring width declined in the 1960s and 1970s, rebounded following high water years in the 1980s, and had a sustained decline after the 1980s that further dropped off in the early 2000s. The decline was not caused by old tree age, but rather other environmental factors. The proportion of living canopy in a tree was consistent with its annual tree ring growth rate, with less live canopy corresponding to smaller tree rings. The dead DW trees were the slowest growing among the six canopy classes for the entire pipeline period. Although this trend was not found in *Populus fremontii* that experienced a rapid water table drawdown (Grismer 2018), a review by Cailleret et al. (2017) found ring width to decline before tree mortality in 84% of populations studied. Annual and decadal tree ring growth patterns in living trees closely resemble those of the dead trees, suggesting additional tree mortality may occur in the future if recent hydroclimatic conditions persist.

Reach and sub-reach tree ring growth characterizations revealed variability at multiple scales. There were differences among subreaches in the DW reach, but tree ring growth showed no consistent downstream trend. This suggested that small-scale growing conditions affected tree growth at scales not captured by our reach-level characterizations. At the reach level, the DW reach declined relative to two reference reaches immediately after pipeline installation, and it declined relative to the other two reference reaches beginning in the 2000s. In summary, ring width in the DW reach decreased following flow diversion in 1961 and again during the recent drought beginning in the 2000s. This result is consistent with the tree and branch dieback documented in the same two periods. On the other hand, two of four reference reaches also showed declines since the 1960s, indicating that growth in some or all of these reaches is controlled at least in part by factors other than flow.

5. Stable Isotopes in Tree Rings

5.1 Introduction

Trees form wood out of water and air by assimilating carbon from CO₂ during photosynthesis. Carbon has two stable isotopes, ¹²C and ¹³C, which have 6 protons paired with 6 or 7 neutrons. The isotopes are nearly identical in their chemical properties, but their different atomic masses result in physical and biological processes discriminating against the heavier isotope (McCarroll & Loader, 2004). Two leaf-level processes cause the ratio of ¹³C to ¹²C, expressed as δ¹³C, to differ from that of atmospheric CO₂ (Farquhar et al., 1982). The first process is fractionation through diffusion, where the larger isotope is less able to diffuse through stomata and into the leaf. The second fractionation occurs at carboxylation, where the photosynthetic enzyme rubisco discriminates against ¹³C as the tree turns carbon into plant tissue. These processes deplete δ¹³C in leaf sugars relative to their source CO₂ in the atmosphere, and some of these sugars are then used to create the wood that forms annual growth rings. Factors that affect stomatal conductance and photosynthetic rate, such as drought, affect the stable carbon isotope ratio in tree rings. Therefore, δ¹³C in tree rings indicates tree water stress at the time of carbon capture (McCarroll & Loader, 2004), with δ¹³C being negatively correlated with water availability in dry environments for many tree species including those in the cottonwood family, *Salicaceae* (Chamaillard et al., 2011; Friedman et al., 2019; Rood et al., 2013; Schifman et al., 2012).

Wood is primarily composed of cellulose, hemicellulose, and lignin. The relative proportion of these components changes with tree age, and the components differ in their δ¹³C values (Friedman et al., 2019; Schleser et al., 2015; Wilson & Grinsted, 1977). Because of this, δ¹³C of tree ring wood can vary systematically through time independent of the external growing conditions. By investigating the cellulose component of wood it is possible to isolate environmental signals that are unaffected by tree aging (Friedman et al., 2019).

Stable isotope analyses can be combined with measures of tree ring growth, such as the basal area increment (BAI) that we used in prior analyses, to improve the environmental signals detectable in tree rings (Au & Tardif, 2012; Schollaen et al., 2013). δ¹³C may even provide a more accurate indication of drought than tree ring width. While ring width often reflects drought (Cook et al., 1999; Meko et al., 2007; Schook et al., 2016a; Schook et al., 2016b), it can also reflect temperature (Csank et al., 2016), pathogenic outbreaks (Andersen, 2015), fire (Schweingruber, 2012), and competition (Willms et al., 2006). The fractionation mechanisms that influence tree ring δ¹³C, however, better isolate water availability as the cause of interannual variation. To complement our analyses of tree ring growth, we analyzed the δ¹³C in cottonwood tree rings to search for indications of when and where drought has affected the Snake Creek riparian corridor.

5.2 Methods

5.2.1 Sample preparation

We analyzed stable carbon isotope ratios (δ¹³C) in cottonwood annual growth rings for 1942–2016 (n = 783 tree rings, plus 61 duplicates). We sampled four trees in each of four Snake Creek groups: dead trees from DW and live trees from DW, RU, and RD. The live and dead DW trees formed a

composite DW group, and the RU and RD trees formed a composite reference group. The selected trees had clearly delineated annual rings, relatively sensitive annual ring widths with at least moderately high correlations with the reach chronology, and survived to at least 2013 (Schollaen et al., 2013). The trees all had pith dates of 1852 to 1913, so our 1942–2016 analysis should avoid the juvenile effect of depressed $\delta^{13}\text{C}$ (McCarroll & Loader, 2004) that has been shown for the first 26 years of life for *Populus deltoides* (Friedman et al., 2019).

Sample preparation was modified from Friedman et al. (2019). Wood samples were collected with a 12 mm diameter increment borer from live trees or cut from cross sectional slabs from dead trees. We excised the entire width of each ring with a razor blade and discarded the glued bottom and sanded top surfaces to avoid contamination. We combined equal weights from two cores from one tree to produce a 4.5–5.5 mg sample. Every year was processed separately for every DW reach tree. For RU and RD, in 4 out of 5 years equal mass was cut from 8 cores from the 4 trees and pooled into an annual sample (Treydte et al., 2007). For every 5th year, the four trees were processed separately to characterize variation among trees.

Wood samples were pulverized to a powder using a Mini-beadbeater-16 (Bio Spec Products Inc., USA) using 2 ml tubes containing a steel bead. Because $\delta^{13}\text{C}$ ratios in cellulose, but not in whole wood, are independent of age in *Populus* (Friedman et al., 2019), cellulose was isolated following methods of Brendel et al. (Brendel et al., 2000). Samples were rinsed with sodium hydroxide to eliminate hemicellulose (Gaudinski et al., 2005), and we added two extra rinses with deionized water to prevent contamination by acetyl groups (Anchukaitis et al., 2008). 0.75–0.85 mg of cellulose was placed in a 4 x 6 mm tin cup (Costech Analytical, Inc., USA). $\delta^{13}\text{C}$ was analyzed using continuous flow isotope ratio mass spectrometry using a Carlo Erba NC1500 interfaced to a Micromass Optima mass spectrometer at the USGS Stable Isotope Laboratory in Denver, Colorado. Stable carbon isotope data are reported as $\delta^{13}\text{C}$ in parts per thousand (‰):

$$\delta^{13}\text{C} = \left(\frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right) * 1000$$

where R_{sample} and R_{standard} are the $^{13}\text{C}/^{12}\text{C}$ ratios in our tree ring samples and the standard, Vienna Pee Dee belemnite (VPDB). We normalized stable carbon isotope data to VPDB using internal lab standards ($\delta^{13}\text{C}$ =−24.60‰ for Holly sugar; −10.82‰ for CNH sugar; and −8.90‰ for USGS glutamic acid 2). We corrected for changing $\delta^{13}\text{C}$ atmospheric carbon dioxide concentrations caused by global fossil fuel emissions using annual corrections developed by McCarroll and Loader (2004) for 1942–2003. These were updated for 2004–2016 using a linear regression of annual averages from the South Pole dataset (NOAA/ESRL/GMD CCGG cooperative air sampling network). Sample reproducibility was evaluated by processing two samples from 61 (7.8%) of the sampled tree rings. 41 of 61 (67%) of the replicates were within 0.20‰ of each other, and 57 of 61 (93%) were within 0.40‰.

5.2.2 Data analysis

Individual trees varied in their average $\delta^{13}\text{C}$ values through time. For example, DW072 averaged −23.8‰ and DW131 averaged −26.40‰. To facilitate comparisons, we analyzed relative $\delta^{13}\text{C}$

changes through time by defining 1942–1961 as the baseline period (McCarroll & Loader, 2004) and evaluating differences relative to this baseline using a t-test.

A k-means cluster analysis was used to group the 16 trees based on their deviation from the standardized $\delta^{13}\text{C}$ values through time. Analysis was performed on the years with values for all trees, which were the unpooled years for reference sites, or every fifth year between 1945 and 2015. Analyses were performed with the kmeans function in the R stats package, and we calculated the CH index to identify the optimal number of groups.

Seasonal correlation analyses were conducted between tree ring $\delta^{13}\text{C}$ and monthly streamflow and precipitation using the seacorr function in the dplR library (Bunn et al., 2019; Meko et al., 2011). Data were from the Great Basin National Park weather station (GHCND: USC00263340, USC00264514) and Cleve Creek stream gauging station (USGS 10343700), as described in Sections 2.3 and 2.4. The duration of the flow record restricted the analyses to 1962–2016. We identified correlations between $\delta^{13}\text{C}$ or tree ring width with flow and precipitation for 1, 3, and 12-month periods. Splined ring width indices were used because they help isolate high frequency variation such as differences in weather between years, and filter out some low-frequency trends such as multi-decadal climate trends. Methods are further explained in Section 6.2.

5.3 Results

5.3.1 $\delta^{13}\text{C}$ through time

Annual tree ring $\delta^{13}\text{C}$ values showed evidence of water shortage following diversion in the DW reach compared to the reference reaches. $\delta^{13}\text{C}$ values increased (i.e., became less negative) by an average of 1.00‰ from the pre- to post-pipeline periods in the DW reach, indicating water shortage. In contrast, $\delta^{13}\text{C}$ values remained similar at RD (+0.03‰) and slightly decreased at RU (−0.38‰; Table 5.1; Figure 5.1) from the pre- to post-pipeline periods. Although the reference reaches differed from each other, their post-pipeline changes were more similar to each other than to the DW dead and live trees, and their trends did not indicate increasing water stress like DW's did. In the pre-pipeline (1942–1961) period, combined DW live and dead trees had mean annual values of −26.6‰ to −25.7‰. By 1964, $\delta^{13}\text{C}$ increased well above the pre-pipeline range to −24.7‰. During most of the post-pipeline period $\delta^{13}\text{C}$ was −25.8‰ to −24.7‰, reaching −23.7‰ in the late 1970s. The DW reach's annual $\delta^{13}\text{C}$ values exceeded their pre-pipeline baseline for the entire post-pipeline period except the wet period of 1982 to 1986 (Figure 5.2; Figure 2.8). In contrast, RD trees remained near their baseline throughout the post-pipeline period, and RU trees dropped slightly below their baseline (Table 5.1). $\delta^{13}\text{C}$ variation in living and dead DW trees was similar through time, indicating similar access to water and water stress. Their similarity, along with increasing $\delta^{13}\text{C}$ values through time, supported interpretations from tree ring chronology analysis that surviving trees in the DW reach closely resemble those that have died (Section 4.3).

The $\delta^{13}\text{C}$ patterns of 16 study trees were best separated into 2 evenly sized groups based on a cluster analysis. All DW trees formed one group and all RU and RD trees formed the other (Figure 5.3). RU and RD trees being more similar to each other than to the geographically intermediate DW trees provided strong evidence that $\delta^{13}\text{C}$ deviated from natural conditions for trees adjacent to the pipeline.

Table 5.1. Description of the $\delta^{13}\text{C}$ isotope samples. R_{bt} is mean between-tree correlation, and EPS is the expressed population signal. Asterisks denotes a significant change at $p < 0.05$ for t-tests of between-period means.

Measurement	DW Live	DW Dead	DW All	RU	RD	Ref. All
Trees sampled (n)	4	4	8	4	4	8
Wood in each sample was from:	2 cores per tree	2 cores per tree	2 cores per tree	8 cores from 4 trees; every 5 th year unpooled at 2 cores per tree	8 cores from 4 trees; every 5 th year unpooled at 2 cores per tree	16 cores from 8 trees; every 5 th year unpooled at 2 cores per tree
Interseries correlation	0.25	0.30	0.35	0.47	0.20	0.21
R_{bt}	0.40	0.33	0.31	0.37	0.14	0.15
EPS	0.71	0.65	0.76	0.70	0.39	0.66
1942–1961 Mean $\pm$ SD	-26.25 ± 0.42	-26.31 ± 0.44	-26.28 ± 0.23	-25.83 ± 0.49	-25.43 ± 0.31	-25.63 ± 0.29
1962–2016 Mean $\pm$ SD	-25.09 ± 0.63	-25.45 ± 0.73	-25.28 ± 0.58	-26.20 ± 0.47	-25.40 ± 0.41	-25.80 ± 0.37
1942–1961 to 1962–2016 change	1.16*	0.87*	1.00*	-0.38*	0.03	-0.17

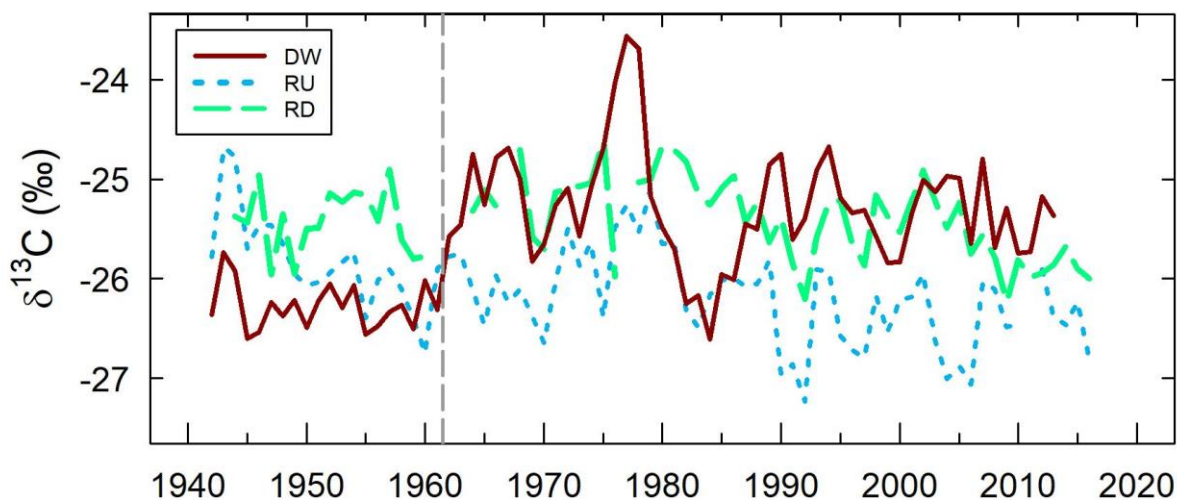


Figure 5.1. Annual tree ring $\delta^{13}\text{C}$ values from DW (live and dead), RU, and RD study reaches for 1942–2016. The vertical dashed line shows the beginning of stream dewatering. Note the 1960s upward shift for DW trees only.

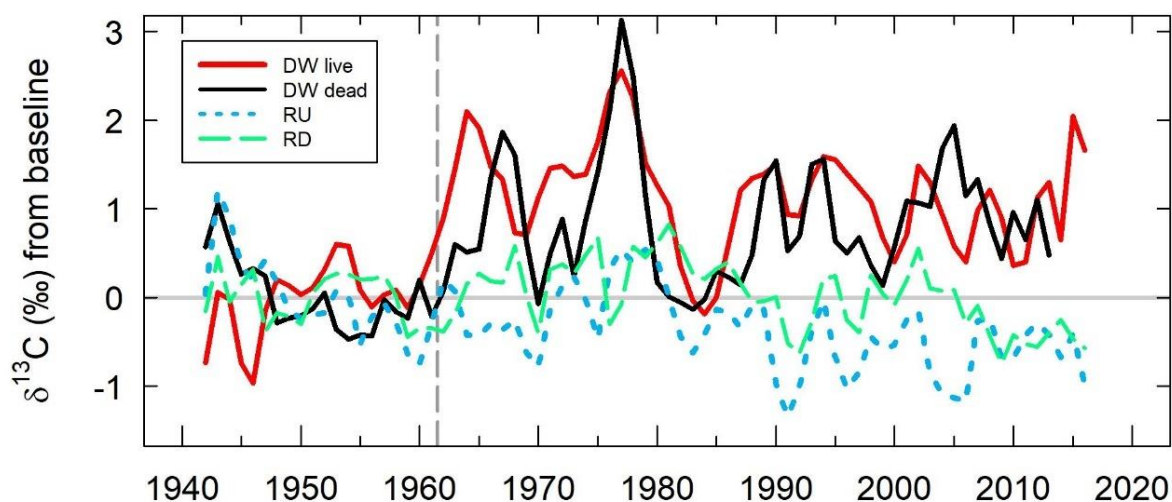


Figure 5.2. Annual tree ring $\delta^{13}\text{C}$ relative to baseline for DW live, DW dead, RU, and RD trees ($n = 4$ each). The y-axis is standardized so the 1942–1961 baseline period averages 0. The vertical dashed line shows the beginning of stream dewatering. Note the 1960s upward shift for DW trees only.

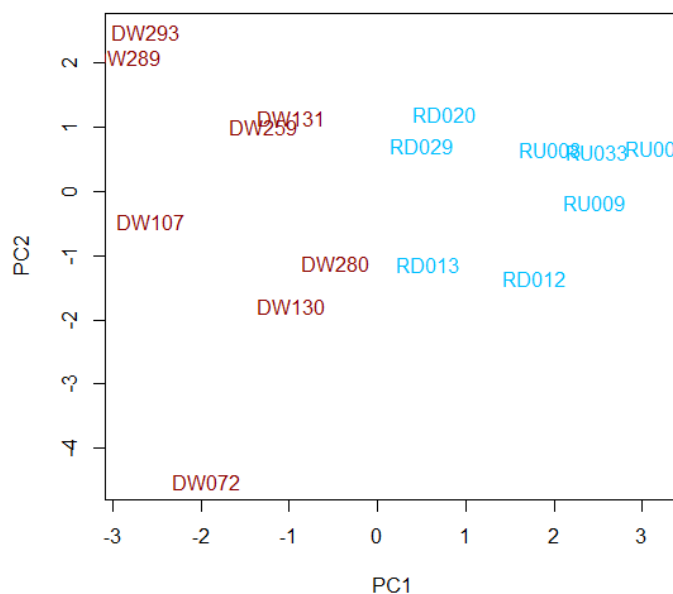


Figure 5.3. Trees plotted on the first two principal components of a k-means cluster analysis based on $\delta^{13}\text{C}$ of tree rings. The 16 trees fell into two evenly sized groups, with all DW trees in one group (red) and all RU and RD trees in the other (blue).

5.3.2 Climate connections to $\delta^{13}\text{C}$

The DW and reference reaches had contrasting relationships of tree ring $\delta^{13}\text{C}$ to flow and precipitation. The DW reach $\delta^{13}\text{C}$ was significantly negatively correlated to flow for 9 of the 14 months tested (Figure 5.4a). This relationship was not significant for any month at RU and it was

positively correlated for 3 months at RD. These null and positive relationships indicated that stomatal closure was less sensitive to flow variability in reference reaches. This would be expected in areas where trees have enough water that growth is limited by some other factor, as likely occurred in reaches with perennial Snake Creek flow.

Tree ring $\delta^{13}\text{C}$ was negatively correlated with precipitation and flow in the DW reach, confirming that drier conditions corresponded to higher $\delta^{13}\text{C}$. The strongest 1-month correlation between $\delta^{13}\text{C}$ and flow were for February and April of the year of tree ring formation ($r = -0.50$ and -0.51 ; Figure 5.4a). The 1-month flow correlations varied little between months because flow is highly correlated among months. 1-month $\delta^{13}\text{C}$ correlations with precipitation were more variable and weaker than correlations with flow (Figure 5.4b). $\delta^{13}\text{C}$ correlations averaged around $r = -0.4$ for both flow and precipitation for 12-month periods (Figure 5.4c,d). This is likely because flow and precipitation are more strongly correlated to each other and representative of the water encountered by riparian trees at longer time steps. $\delta^{13}\text{C}$ values were unexpectedly slightly more related to precipitation from the year prior to tree ring formation than the year of formation (pink in Figure 5.4d). This could be because deciduous trees use carbohydrates from the previous year to make the first cellulose of the spring. The previous year's growing conditions affect both the quantity of and isotopic ratios in stored carbohydrates, which could have caused the stronger connection to the prior year for $\delta^{13}\text{C}$ compared to ring width (Friedman et al., 2019). In contrast to $\delta^{13}\text{C}$, tree ring widths were more strongly correlated with current spring and summer's flow and precipitation (also see Section 6.3). This differed from *Populus deltoides* in North Dakota that had a higher correlation of $\delta^{13}\text{C}$ with precipitation in the summer of tree ring formation than in prior years (Friedman et al., 2019). Our relatively weak relationships between $\delta^{13}\text{C}$ and precipitation could reflect the patchiness of precipitation in the basin and range physiographic province, as the weather station used was ~10 km away and may not adequately reflect precipitation in the study reaches. Overall, $\delta^{13}\text{C}$ generally had weaker correlations than ring width with flow and climate, in part reflecting the larger sample size for ring width ($n = 116$ trees) than for $\delta^{13}\text{C}$ ($n = 8$ trees).

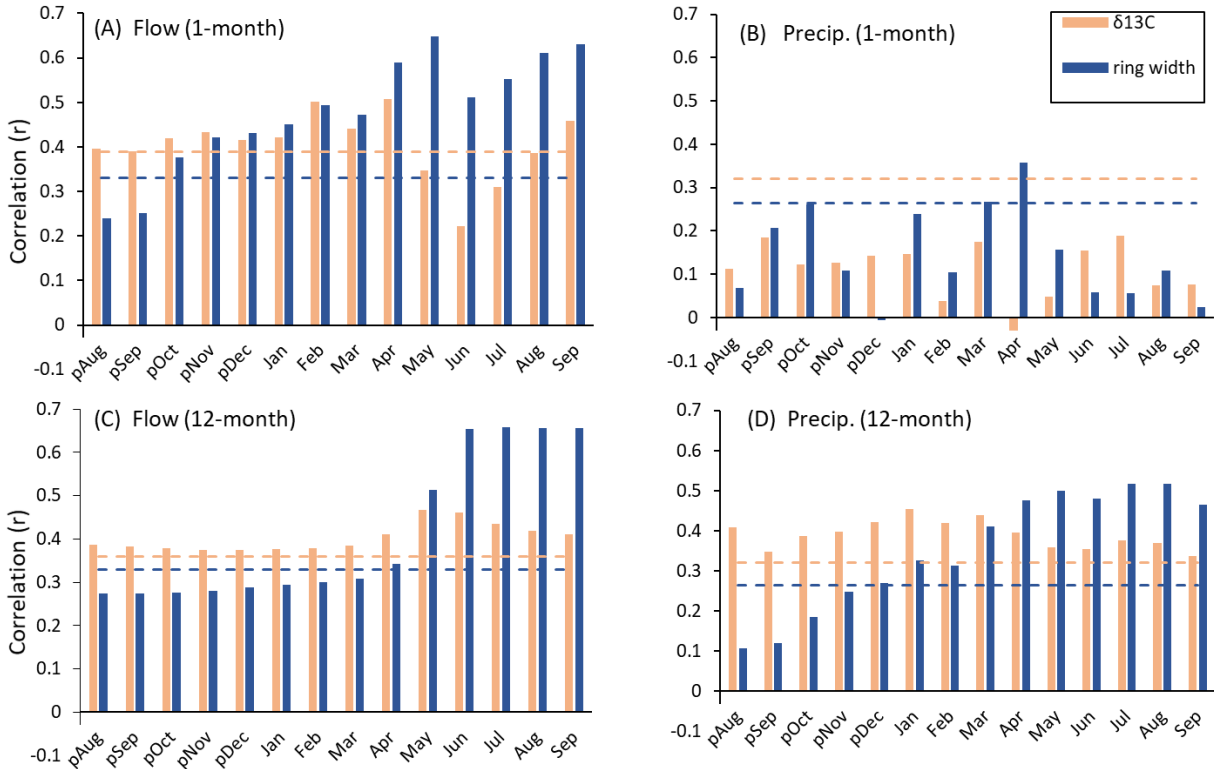


Figure 5.4. Correlations for tree ring width or $\delta^{13}\text{C}$ with flow and precipitation in the DW reach (1962–2016). Bars are $\delta^{13}\text{C}$ (pink) and ring width (dark blue) correlations with 1-month (A, B) or 12-month averages (C, D) for flow (A, C) and precipitation (B, D). $\delta^{13}\text{C}$ correlations are shown as inverses of their true values to facilitate comparison. 12-month data are averaged over the 12 months ending with that labeled; for example, “Sep” correlations are for previous October (pOct) through current September. Dashed lines show approximate 95% significance thresholds.

5.3.3 Relationships between tree ring width and $\delta^{13}\text{C}$

In the post-diversion period annual tree-ring $\delta^{13}\text{C}$ in the DW reach was negatively correlated with BAI for both live ($r = -0.41$) and dead ($r = -0.29$) trees (Table 5.2), consistent with the hypothesis that growth was limited by water availability. In the pre-diversion period these negative correlations were weaker ($r = -0.34$ and $r = -0.05$), confirming that water limitation was not as strong prior to diversion. In contrast growth in the reference reaches was non-significantly positively correlated with $\delta^{13}\text{C}$ in both the pre- and post-diversion periods (Table 5.2) indicating that growth was limited by some factor other than water limitation.

Table 5.2. Correlations (r) between $\delta^{13}\text{C}$ and basal area increment (BAI) for the pre-pipeline (1942–1961) and post-pipeline (1962–2016) study periods across study reaches. Asterisks denote a regression slope differing from zero (* if $p < 0.05$ and ** if $p < 0.01$.).

Group	Correlation (r)	
	Pre-pipeline (1942–1961)	Post-pipeline (1962–2016)
DW Live	−0.34	−0.41**
DW Dead	−0.05	−0.29*
DW All	0.00	−0.39**
RU	+0.25	+0.20
RD	+0.34	+0.03
Reference All	+0.07	+0.17

5.4 Discussion

Stable isotope analyses provided a clearer indication of water stress in the DW reach trees than did annual tree ring width. Findings were consistent with the mechanistic understanding that drought induces stomatal closure, which slows diffusion of CO_2 into the leaf, resulting in depletion of ^{12}C within the leaf, leading to ^{13}C enrichment (McCarroll & Loader, 2004; Schifman et al., 2012). Cottonwoods from the DW reach had a distinct and largely sustained increase in $\delta^{13}\text{C}$ beginning in the early 1960s, at the onset of stream dewatering. There was no concomitant change in trees from either reference reach, suggesting that stomatal closure to lower transpiration rates changed abruptly only at DW. A cluster analysis split the 16 trees into two groups that were perfectly separated into DW (live and dead) and reference (RU and RD) groups. Thus, even with relatively high tree-to-tree variability there was a clear reach-level signal. The difference cannot be attributed to climate differences because the DW reach is located between the two reference reaches.

In riparian environments of the North American Great Plains, *Populus* have been shown to have higher $\delta^{13}\text{C}$ in years of low stream flow (Friedman et al., 2019; Rood et al., 2013). Our analyses indicate that a transition occurred in the early 1960s that initiated continuous drought conditions in the DW reach. $\delta^{13}\text{C}$ returned to baseline levels only in the 1980s, a wet period when Snake Creek discharge likely exceeded the pipeline’s intake capacity and surface water would have flowed through the DW reach. In general, the half-century of mostly sustained high $\delta^{13}\text{C}$ levels indicated prolonged water stress capable of tree branch and whole tree mortality (Allen et al., 2015).

The relationship between $\delta^{13}\text{C}$ and BAI (i.e., ~ring width) differed for trees in DW compared to reference reaches. They were negatively correlated under the dry conditions in the DW reach, supporting the other analyses we have presented that reveal persistent water stress in the reach. In contrast, the reference trees had weakly *positively* correlated $\delta^{13}\text{C}$ and BAI, suggesting their growth was not limited by water scarcity (Warren et al., 2001). Seasonal correlation analyses supported this conclusion, with weak connections between $\delta^{13}\text{C}$ and water variables, and a slight *positive* correlation

between $\delta^{13}\text{C}$ and flow at RD (data not shown). Given these spatial and temporal patterns, we conclude that the drought-signals contained in $\delta^{13}\text{C}$ and BAI provide complementary evidence that Snake Creek cottonwoods in the DW reach have uniquely experienced persistent drought since the 1960s.

6. Hydroclimatic Drivers of Tree Ring Width

6.1 Introduction

Cottonwood growth is influenced by changes in the local growing environment both within and between years. Flow, precipitation, and temperature are all potential driving variables for photosynthesis, tree ring production, and tree health (Schook et al., 2016a). Understanding the relative influence of these hydroclimatic variables in GRBA could help reveal if the Snake Creek riparian forest is dependent on water access. Monthly or seasonal connections to these driving variables often reveal stronger relationships than annual averages (Cook et al., 2013; Meko et al., 2015; Schook et al., 2016b), which informs resource management. Considering that precipitation and temperature are regional processes beyond our control, gaining a better understanding of the roles of each driving variables could influence streamflow management actions in GRBA.

6.2 Methods

We investigated relationships between tree ring width and stream flow, precipitation, and temperature to identify the hydroclimatic drivers supporting tree ring production. For each tree, annual ring widths were regressed against 1, 3, and 6-month averages of flow, precipitation, and temperature. We tested correlations for 14 periods having ending months between the previous year's August and the current year's September. For example, the 3-month February temperature correlation is for the average temperature between the previous year's December and current February. We present results for 3-month averages only because they isolate seasons and were relatively strong. We compared correlation coefficients (r) for trees growing in the dewatered reach (DW) to those from the four reference reaches (RU, RD, RB and RS) for 1962–2016. Analyses were performed using the `seascorr` function in the `treeclim` package in R (Bunn et al., 2019; Meko et al., 2011). Ring widths were detrended using a spline with 75% frequency response cutoff at 67% of the series length. Splined tree ring width indices were selected instead of BAI used in tree ring chronologies because the relatively flexible splines removed low-frequency trends and improved comparisons to high-frequency weather variation. Cleve Creek data were used for flow because the record was much longer than Snake or Baker Creeks, and Cleve and Snake Creek monthly flows correlated at $r=0.89$ (see Section 2.4). We did not attempt to quantify flow for the DW reach specifically because clogging of the pipeline inlet, leaks in the pipeline, and groundwater resurgence made its discharge variable through space and time.

In a second analysis we investigated pre- (1939–1961) and post-pipeline (1962–2016) differences in tree ring width correlations with precipitation and temperature. We did not use flow because the record was too short. Between-period differences at DW could have been due to stream dewatering, but differences in other reaches were representative of reference conditions. We interpreted pipeline-induced effects by evaluating DW changes compared with reference reach changes.

We also analyzed individual trees in DW to characterize tree-to-tree differences. We performed the above 3-month correlation analysis for 1962–2016 on 116 living trees. For each tree we identified the ending month with the strongest correlation to flow, precipitation, and temperature. We summed the

number of trees with peak correlations in each month to assess their distribution through time and evaluate the variability within the reach.

6.3 Results and Discussion

6.3.1 Tree rings and flow, precipitation, and temperature

Tree ring width was more strongly correlated to flow than to precipitation or temperature (Figure 6.1). This was strongest at the DW reach, which had a maximum correlation for the 3-month period ending in May ($r = 0.66$). This exceeded the reach's strongest correlations with precipitation (May, $r = 0.43$) and temperature (April, $r = -0.35$; Figure 6.1a). The DW peak correlation to flow ($r = 0.66$) was also higher than its peak for the four reference reaches ($0.33 < r < 0.52$). Only in the DW reach did tree ring width and flow have significant positive correlations for every 3-month period between the previous November and current September. The 1962–2016 peak annual flow occurred most often in May (26 of 52 years). Years of high snowmelt runoff appear to have been important for GRBA cottonwood growth.

Tree ring width was more strongly related to precipitation for trees at DW compared to the reference reaches, with correlations being the highest at DW for 12 of the 14 periods tested (Figure 6.1b). This correlation was highest for DW trees in May ($r = 0.43$), the same month the flow relationship peaked. Precipitation and flow are mechanistically related and positively correlated, but differences between them are caused by evapotranspiration and discharge delays from snowpack storage and runoff pathways. In addition, precipitation in our study reaches differed from that in the mountains where flow was generated. Partial correlations (data not reported) revealed both metrics to be independently and significantly related to tree ring growth. The strength of flow and precipitation correlations with ring width indicated that cottonwood growth was most water-limited at DW compared to the reference reaches. For all reaches, flow was a better predictor of tree ring width than was precipitation, likely because cottonwood roots are distributed to respond to a rise in alluvial groundwater more than surface soil wetting (Rood et al., 2011; Williams & Cooper, 2005). The DW reach likely had the strongest correlations because water access became more variable and limiting under stream dewatering.

Temperature was negatively correlated to tree ring width, but the correlation was weaker than the positive relationships with flow and precipitation (Figure 6.1c). The strength of the DW temperature relationship was intermediate among the five study reaches, and the three months ending in April ($r = -0.35$) and May ($r = -0.31$) were the only significant periods. Across all reaches, the only significant temperature correlations were for 3-month periods ending between March and June, indicating that higher winter and spring temperatures reduced tree ring width. During warm winters and springs snow melts earlier, and more snowmelt is likely to runoff before trees emerge from dormancy and begin photosynthesis.

6.3.2 Comparing pre- and post-pipeline periods

Comparing the pre- and post-pipeline periods, the correlation between tree ring width and precipitation increased in the DW reach but generally decreased at the four reference reaches (Figure 6.2). DW tree ring width was significantly related to precipitation for three of the 3-month pre-pipeline periods, and six periods post-pipeline. The strongest relationships were for post-pipeline

periods ending in April and May ($r^2 = 0.43$). Although the pre-diversion flow record was too short to analyze, precipitation generates streamflow and both sources provide water for riparian trees. These findings support the idea that DW tree growth was highly correlated to flow in the post-pipeline period, when water was likely scarcer. The post-pipeline period had high variability in flow between years and many years had little-to-no flow in the DW reach. Because the pipeline can only divert base flow and small floods, years with large snowpacks and precipitation events created sufficient runoff-driven flow to surpass the pipeline capacity. This created occasional streamflow through DW capable of replenishing alluvial groundwater.

Temperature relationships were similar among DW and reference reaches, and all correlations to ring width were generally negative and small in the pre-diversion period. Correlations at DW were intermediate among the five reaches (Figure 6.2c,d). The strongest correlations were significant negative connections to temperature for April at RS, RU, and DW during the post-pipeline period (Figure 6.2d). Across all reaches, there was a small increase in sensitivity to temperature variability in the post-pipeline period. In an analysis not included in this report, we found ring width from older trees to be more reduced by high temperature compared to young trees, so the difference between periods was consistent with the aging trees and riparian forest.

6.3.3 Individual trees

Ring width from individual trees in the DW reach was most strongly related to spring conditions for flow, precipitation, and temperature. The 3-month periods ending in April and May of the current year had the strongest relationships, with one of these periods having the highest correlation with flow for 47% of trees, precipitation for 42% of trees, and temperature for 43% of trees (Figure 6.3). Individual trees had their highest correlations in nearly every period analyzed, highlighting that each tree had distinct growth patterns likely due to microhabitat and resource availability.

The highest ring width correlations most commonly occurred for the three months ending in April for precipitation and temperature, and in May for flow. The one-month lag for flow was likely related to Snake Creek discharge being generated from snowpack that melts with increasing spring air temperature. Runoff from high-elevation snowpack enters Snake Creek and supports cottonwoods, but warmer springs reduced growth. This could be because February–April temperatures affect whether precipitation falls as snow or rain and the timing of snowmelt runoff. Maintaining high-elevation snowpack through the spring preserves more runoff for the growing season when it is accessible to trees. Alternatively, warmer spring temperatures could cause trees to emerge from dormancy early and leave them susceptible to damaging frosts, which might also reduce tree ring growth. Hotter and drier conditions will increase the water-stress already present, and diverting water from streams is likely to exacerbate this.

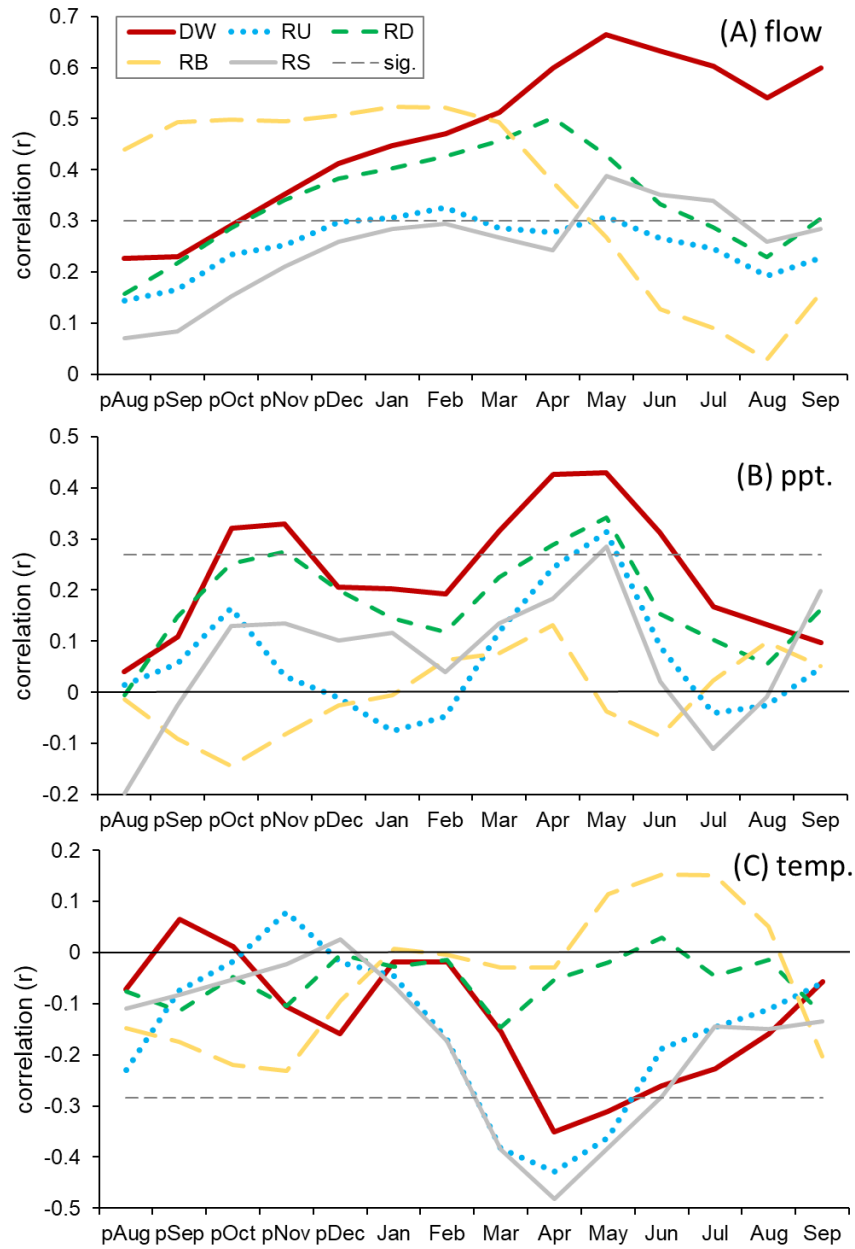


Figure 6.1. Tree ring correlations to flow (A), precipitation (B), and temperature (C). Values are 1962–2016 correlations between annual ring width and 3-month-averaged weather, where “p” is for the previous year (e.g., Feb flow correlation is for average flow in pDec–Jan–Feb). Note that y-axes differ, and panels have a black line at $y = 0$. The dashed grey line is the approximate level of significance.

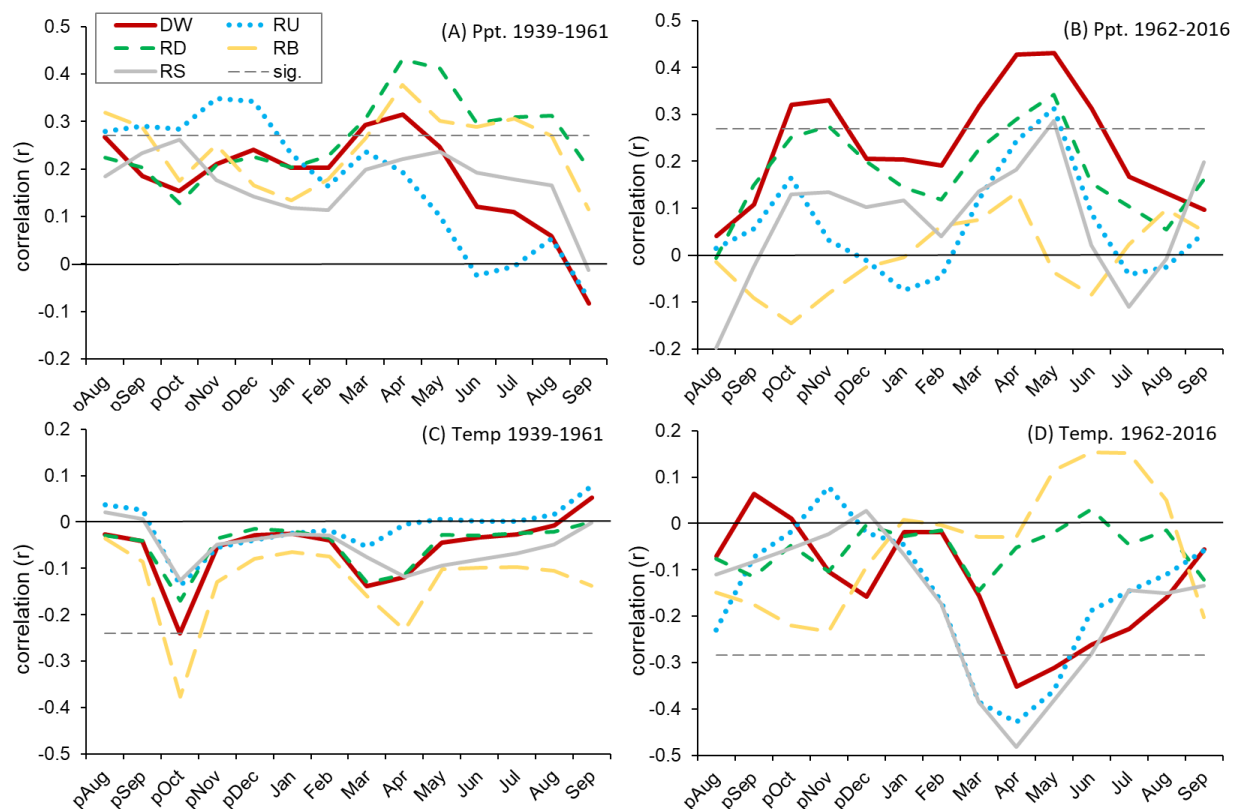


Figure 6.2. Annual tree ring width correlation to precipitation (A, B) and temperature (C, D) before (A, C) and after (B, D) the pipeline was installed. Values are correlations between annual ring width and 3-month-averaged weather, where “p” is for the previous year (e.g., Feb flow correlation is for average flow in pDec–Jan–Feb). Note that y-axes are constant within variables but different between them. Panels have a black line at $y = 0$ and dashed grey line at the approximate level of significance.

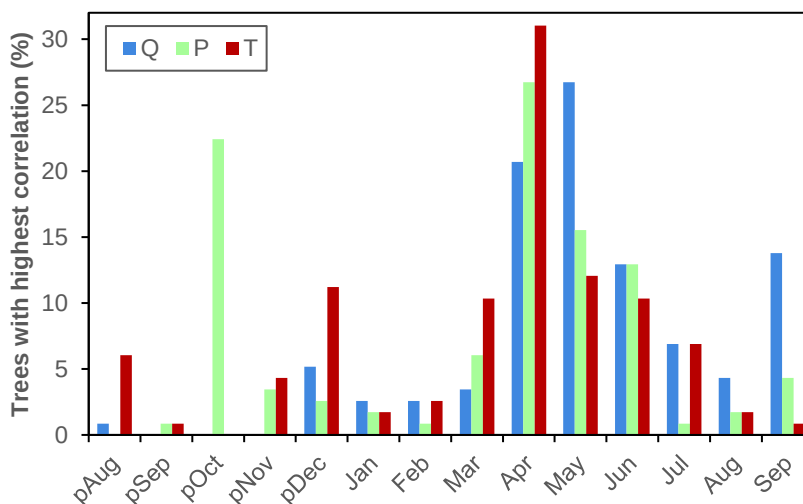


Figure 6.3. Percent of DW reach trees (n = 116) with maximum correlations between ring width and flow (Q, blue), precipitation (P, green) and temperature (T, red) for 3-month period ending in the x-axis month. Note that all correlations were positive for flow and precipitation but negative for temperature.

7. Cottonwood Ecophysiology

7.1 Introduction

The canopy declines documented in previous sections resulted from trees being unable to maintain their biological structure under the modern growing conditions (Jump et al., 2017). Investigations of tree physiological processes can be used to explain the mechanisms underlying tree functioning and forest decline (Cooper et al., 2003; Kaluthota et al., 2015; Rood et al., 2000). For example, following drought-induced branch dieback, *Populus trichocarpa* have been shown to rapidly regain branch growth and physiological function after the hydrologic connection between the stream and riparian groundwater was restored (Amlin & Rood, 2003). Increasing our understanding of the processes underlying Snake Creek cottonwood declines could help identify requirements for maintaining the riparian forest, including identification of resource needs such as the flows required for tree growth and persistence (Stromberg & Patten, 1990). Clarifying the mechanisms contributing to riparian forest decline along Snake Creek could inform park managers and help facilitate recovery of the riparian forest.

A tree's water balance is regulated by ecophysiological processes. Water retention is vital to plant survival, and xylem water potential measurements are commonly used to evaluate plant water stress. Xylem pressure varies daily and seasonally with variations in sunlight, evaporative demand, and water access. Different tree species have evolved drought avoidance or resistance strategies through physiological mechanisms, including stomatal regulation to limit transpirative water loss (Attia et al., 2015; McDowell et al., 2008). Riparian *Populus* most likely avoid, instead of resisting, drought by closing their stomata in the warmest portion of each day, as indicated by their high vulnerability to xylem cavitation (Stewart Rood, *pers. comm.*, 27 Apr 2019). Stomatal regulation reduces transpiration, but also reduces total daily photosynthesis. This makes drought-avoiding trees (i.e., isohydric species) susceptible to carbon starvation if their total photosynthesis cannot support the metabolic demands of their living tissues (McDowell et al., 2011).

To investigate the ecophysiology of Snake Creek cottonwoods, we compared leaf water potential between reaches during predawn, midday, and repeatedly throughout the day. We also experimentally measured the ability of leaves to transport water as a function of decline in xylem water pressure. The goal of this research was to identify the physiological causes of cottonwood decline to better inform management of the Snake Creek riparian ecosystem.

7.2 Methods

7.2.1 Field-based leaf water potential

We measured leaf water potential during predawn (ψ_{pd} ; 04:30–06:30) and midday (ψ_{md} ; 12:30–14:30) hours for cottonwoods in the dewatered (DW) and reference downstream (RD) reaches. Sampled trees in the DW reach averaged 24% live canopy and RD trees averaged 74% live canopy, similar to their reach averages (Figure 3.7). Metal tags were placed on the north side of all trees 1.5 m above the ground.

We measured 10 trees located within 100 m of each other in each reach to enable fast processing of the time-sensitive data. Terminal branches 5–6 m above ground were cut, collected, and immediately sealed in a plastic bag containing a damp cloth. Midday leaf water potential was measured the following day from 12:30–14:30 under full sun. Three leaves from each branch were measured within 10 minutes of cutting using a Scholander-type pressure chamber (PMS Instruments), and we found high within-branch consistency. T-tests were conducted to identify differences between reaches for different times of day by comparing the mean value of the three leaves from every sample branch (Table 7.1).

Diurnal trajectories of leaf water potential were created using the subset of 5 trees from each reach that had the greatest number of branches accessible at sampling height. Repeated measurements were taken every half-hour between 06:00 and 13:00. We used a mixed-effects analysis of variance to identify if there was an interaction effect between the within-subjects variable of time and the between-subjects variable of reach. We then used the least squares means test with no multiple comparisons correction factor to identify at which times of day leaf water potentials differed between reaches. We measured one leaf per tree for each sample period because values generally differed by < 0.1 MPa among 3 leaves from a sampled branch.

7.2.2 Percent loss of leaf hydraulic conductance

To experimentally measure leaf hydraulic conductance under water stress, we conducted leaf-cut rehydration experiments using a 60–80 cm long branch from 7 trees in each study reach (Schulte et al. 1987; Table 7.1). Branches were placed in a bucket of river water, covered with plastic, and allowed to rehydrate overnight (Gleason et al., 2014). The following morning, maximum leaf hydraulic conductance (K_{leaf}) and the percent loss of K_{leaf} per unit decline in leaf water potential (i.e., percent loss of conductance, PLC) were measured using the rehydration kinetics method (Brodribb & Cochard, 2009; Brodribb & Holbrook, 2003). We removed two leaves of equivalent shape, size, and height on the branch and dried them for 0–4 minutes. The first leaf was immediately placed in a sealed plastic bag to stop water loss, while the water potential of the second leaf was immediately measured in the pressure chamber. The first leaf was then illuminated by the midday sun, the petiole placed under water, re-cut with a razor blade, and the leaf allowed to rehydrate for 5–45 seconds, the duration of which was informed by the original water potential (ψ_o) of the second leaf. The water potential of this semi-rehydrated leaf was then measured to determine the final water potential (ψ_f). We repeated this procedure with 25–30 pairs of leaves per reach to construct a leaf hydraulic conductance curve using the equation (Brodribb & Holbrook, 2003):

$$K_{leaf} = -\ln\left(\frac{\psi_f}{\psi_o}\right)\frac{C}{t}$$

Where K_{leaf} is leaf hydraulic conductance ($\text{mmol m}^{-2} \text{s}^{-1} \text{MPa}^{-1}$), ψ_f and ψ_o are the final and original leaf water potential (MPa), t is the rehydration time (seconds), and C is leaf capacitance and was estimated separately after Brodribb and Holbrook (2003) as the change in relative water content (mmol m^{-2} leaf area) per unit change in pressure (MPa). Data were fit with a sigmoidal model (% of maximum $K_{leaf} \sim \psi_o$), and the leaf water potentials corresponding to a 12%, 50%, and 88% loss of conductance (P_{12} , P_{50} , P_{88}) were extracted from the curves.

Table 7.1. Summary of ecophysiology measurements made in the dewatered (DW) and reference downstream (RD) reaches in 2017.

Measurement	Trees (n)	Sample unit	DW sampling	RD sampling
Predawn (ψ_{pd})	10	Mean of 3 leaves/tree	Aug 31	Aug 31
Midday (ψ_{md})	10	Mean of 3 leaves/tree	Sep 1	Sep 1
Diurnal (ψ)	5	1 leaf	Sep 2	Sep 1 & 3 ¹
Leaf hydraulic conductivity (K_{leaf})	7	2 paired leaves	Aug 31	Aug 31

¹ RD daily data from 06:00 to 07:30 were from Sept. 3 after data from those times on Sept. 1 were suspect.

7.3 Results

7.3.1 Field-based leaf water potential

We found small yet statistically significant differences in leaf water potentials between the two study reaches. Leaves from DW trees had more negative midday water potentials than RD trees, at -1.61 MPa vs. -1.41 MPa ($t = 2.59$, $p = 0.019$; Figure 7.1). These values were nearly identical to dewatered (-1.55 MPa) and reference (-1.42 MPa) *Populus balsamifera* in Alberta (Amlin & Rood, 2003). Predawn leaf water potentials had the opposite pattern, with trees from DW having higher water potentials compared to RD trees, -0.17 MPa vs. -0.27 MPa ($t = 3.89$, $p = 0.001$), which was similar to *Populus spp.* growing along other western rivers (Leffler et al., 2000; Smith et al., 1991; Williams & Cooper, 2005). Both reach's pre-dawn leaf water potentials were in the low-stress range and indicated that trees recovered from water stress during overnight rehydration. Predawn and midday differences between study reaches were small, and it is unclear if they are ecologically meaningful. Others have found a similar magnitude of differences between and within study reaches, illustrating that local conditions influence water potential measurements (Williams & Cooper, 2005).

Diurnal measurements had higher temporal resolution of leaf water potential and were consistent with the predawn and midday findings. Leaf water potentials were marginally different between reaches ($F = 4.3$, $p = 0.07$), but differed greatly throughout the day ($F = 79.6$, $p < 0.001$). The interaction between reaches and time was also strong ($F = 7.7$, $p < 0.001$). Before sunrise, at 06:00, DW cottonwoods had leaf water potentials averaging -0.24 MPa, while the RD trees which were measured the next morning, and one day more removed from rain, were more negative at -0.54 MPa (Figure 7.2). Xylem potentials reached -1.0 MPa at both sites by 07:30. Leaf water potentials continued to drop until about 10:00, and they remained at similar values through 13:00. This midday plateau occurred at a significantly lower value at DW -1.70 MPa (± 0.18 standard deviation) compared to the RD average of -1.45 MPa (± 0.13) ($t = 5.2$, $p < 0.001$). Leaf water potential within a tree varied by an average of 0.44 MPa during this four-hour midday period. This illustrated that trees opened and closed their stomata throughout the day, which reduced water loss and photosynthetic rate.

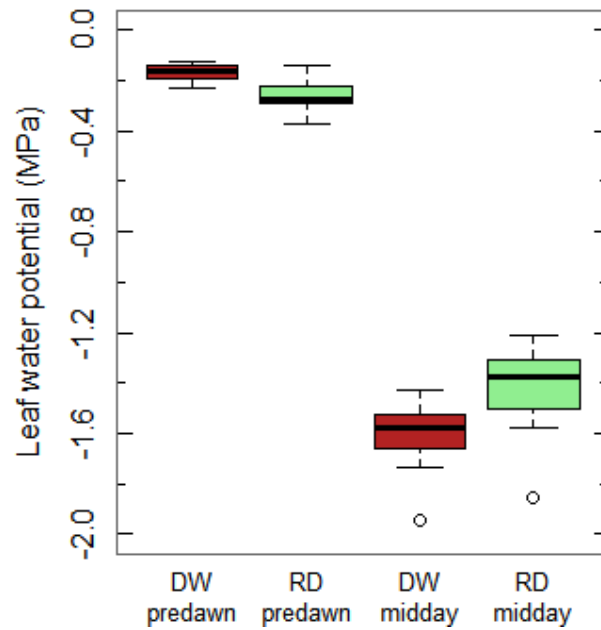


Figure 7.1. Leaf water potentials for cottonwoods in dewatered (DW) and reference downstream (RD) reaches for predawn and midday periods. All four means were significantly different ($p < 0.05$).

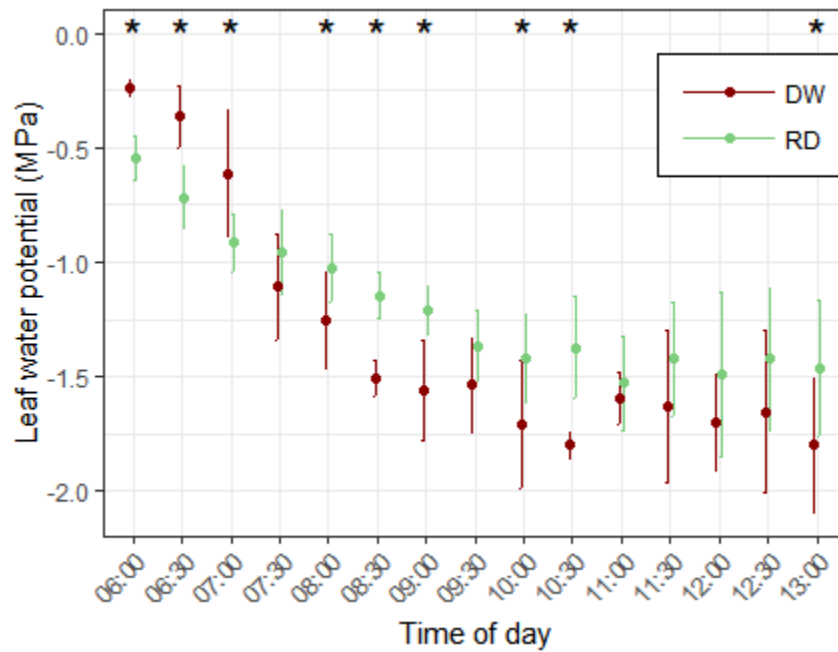


Figure 7.2. Leaf water potentials for cottonwoods in the DW (dark red) and RD (light green) reaches between 06:00 and 13:00. Values are means and 95% confidence bars. Asterisks denote significant differences between reaches ($p < 0.05$).

7.3.2 Leaf hydraulic conductivity

Loss of hydraulic conductivity measurements indicated that trees from the RD reach maintained higher values of K_{leaf} with declining leaf water potential (Table 7.2; Figure 7.3). This suggests that DW trees were more drought avoidant, while RD trees were more drought tolerant, or resilient, under the experimentally dry conditions (Figure 7.3). The maximum K_{leaf} identified using best-fit curves was $47 \text{ mmol m}^{-2} \text{ s}^{-1} \text{ MPa}^{-1}$ at RD and $65 \text{ mmol m}^{-2} \text{ s}^{-1} \text{ MPa}^{-1}$ at DW. The higher value at DW could have been caused by larger or more numerous vessels, although the difference could also be conservatively disregarded as resulting from a single outlier shown as the top-left data point in Figure 7.3. P_{50} was significantly more negative for RD trees (-1.01 MPa) compared to DW trees (-0.69 MPa ; Table 7.2), and P_{88} was also (-1.90 MPa and -1.24 MPa , respectively). These values indicated that RD trees better maintained leaf hydraulic conductivity and continued photosynthesis under drier conditions.

Table 7.2. Percent loss of conductivity for the DW and RD reaches, with means and 95% confidence intervals reported. For example, a leaf in the DW reach lost 50% of its conductivity when dried to -0.69 MPa .

Percent loss conductivity	Dewatered (MPa)	Reference (MPa)
12% (P_{12})	-0.24 (-0.20 – -0.29)	-0.33 (-0.28 – -0.37)
50% (P_{50})	-0.69 (-0.62 – -0.73)	-1.01 (-0.82 – -1.10)
88% (P_{88})	-1.24 (-1.06 – -1.44)	-1.90 (-1.40 – -2.29)

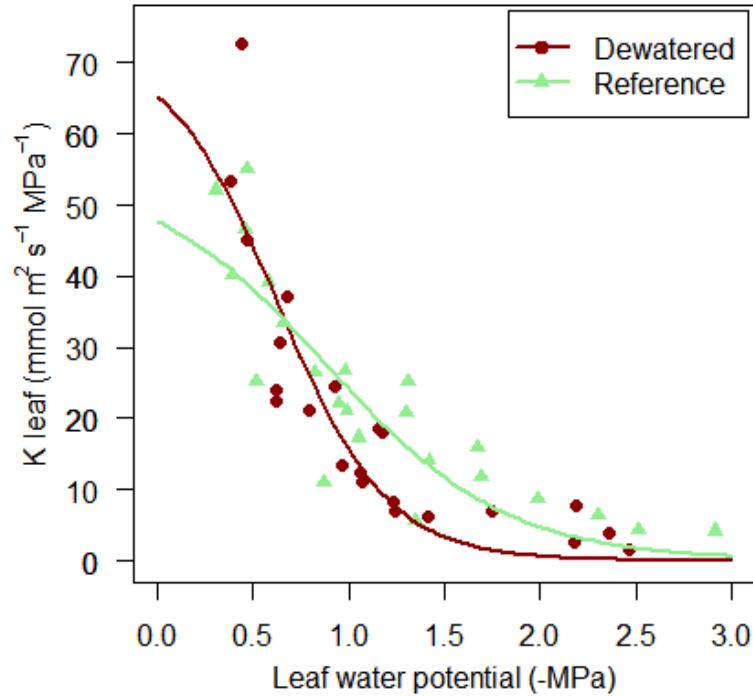


Figure 7.3. Percent loss of hydraulic conductance curves for leaves sampled from the DW (dark red) and RD (light green) reaches. X-axis values are *negative* initial leaf water potentials (Ψ_o).

7.4 Discussion

Cottonwood leaf water potentials were slightly different in the DW and RD reaches, indicating that local growing conditions had a minor effect on tree water relations. However, the general similarity between reaches suggested that their basal growth rate and branch adjustments allowed trees in both reaches to retain leaf-level function under the growing conditions encountered (Amlin & Rood, 2003; Williams & Cooper, 2005). Trees in the DW reach had over 50 years to adjust to stream dewatering. The higher canopy dieback at DW reduced leaf area per tree, which likely reduced the leaf area-to-xylem cross-sectional area compared to the reference reaches. The lower ratio would cause less transpiration demand per xylem area, allowing the xylem to conduct more water and protect against very low leaf water potentials. These adjustments would have improved the ability of roots to replenish water lost in the canopy.

In contrast to the *P. angustifolia* stem water potentials reported by Tyree et al. (1994), our leaf water potential measurements revealed P_{50} conductance to be much less negative than midday water potentials. However, this finding should not necessarily be interpreted as the leaves entering risky or lethal territory because other mechanisms can protect leaves. In contrast to stem water potentials, it is known that declining K_{leaf} is not caused primarily by embolizing vessels, but rather from resistances beyond the vessels (Scoffoni et al., 2011; Secchi & Zwieniecki, 2010). Future Snake Creek cottonwood analyses could explore additional leaf-level measurements such as stomatal conductance, photosynthetic rate, and leaf mass per leaf area (Kaluthota et al., 2015).

The leaf water potential differences between our two study reaches resembled those of *P. trichocarpa* from the Pacific northwest (Dunlap et al., 1993; Schulte et al., 1987) where coastal cottonwoods from wet growing conditions did not develop stomatal sensitivity to drought, whereas cottonwoods from the drier inland site did (Dunlap et al., 1993; Schulte et al., 1987). This suggests that the coastal trees had adapted to reliable water access, making them more likely to continue photosynthesis through drier conditions. This presumably occurred because reliable water access created safer growing conditions. At Snake Creek, analogous conditions could explain why K_{leaf} was roughly twice as high for RD as DW at relatively low water potentials of -1.0 to -1.6 MPa (Figure 7.3), which were like the values measured during midday. Ultimately, the small water potential differences we identified between reaches suggest that other adjustments, such as those documented in our other analyses, document stronger differences between cottonwoods in the DW and reference reaches.

8. Aerial Photos

8.1 Introduction

Aerial photographs are commonly used to investigate historic conditions of rivers and riparian vegetation (Allen et al., 2018; Cadol et al., 2011; Hajdukiewicz & Wyżga, 2019; Schook et al., 2017). Aerial photos capture landscape-scale conditions at discrete dates, enabling retrospective assessment of landscape conditions. Aerial photo analysis is a remote sensing technique that complements field work because it reveals conditions leading up to those encountered at present. In previous analyses we used diverse field surveys to characterize modern conditions at Snake Creek. Additionally, we used tree ring widths to build perspective of past conditions by analyzing information created by cottonwood physiological and morphological processes. This aerial photo analysis adds to the historical data by providing tree canopy information on past forest conditions.

8.2 Methods

8.2.1 General approach

Aerial photographs from fourteen dates spanning 1946–2017 were downloaded at the highest available resolution from USGS Earth Explorer (Table 8.1). The photos had different spatial resolutions and were resampled to achieve a uniform 2 m² pixel size, which did not adversely affect our interpretation of vegetation characteristics. We georectified all photos from 1946 to 1985 to a root mean squared error of < 3 m, and photos from 1994 to 2017 were pre-georectified. Because the photos had different numbers of spectral bands, they were all standardized to a single grayscale band. All photos are from the growing season, but the day of year varied.

Snake Creek was divided into a dewatered (DW) and reference (R) reach for aerial photo analysis (Figure 8.1). The 5-km DW reach was the same as in other analyses and extended from the Snake Creek pipeline inlet downstream to the pipeline outlet. The R reach included two subreaches, one extending 1.5 km upstream from the DW reach and the other extending 2 km downstream from it. The R reach was nearly twice as long as the combined RU and RD reaches from the cottonwood analyses. This longer distance improved the aerial photo analysis because it easily incorporated additional riparian areas that were not feasible to add in the field study.

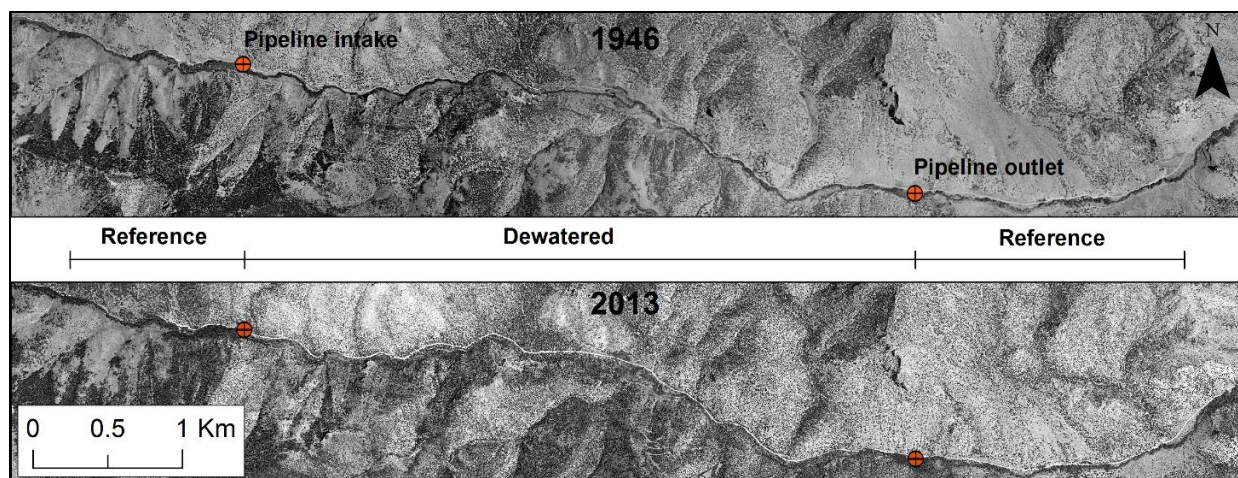


Figure 8.1. The Snake Creek aerial photo study reaches in 1946 (top) and 2013 (bottom). The riparian forest is a dark band, and the road above it is white. Note that in 2013, especially in the DW reach, the distinct riparian forest from 1946 had blended with the upland vegetation after riparian forest dieback and upland forest encroachment.

Table 8.1. Aerial photo metadata including collection year, capture date, spectral bands, collecting agency, and spatial scale. Acronyms: National Agricultural Image Program (NAIP); National Aerial Photography Program (NAPP); Digital Orthorectified Quarter Quads (DOQQ); Red-Green-Blue (RGB).

Year	Capture Date	Spectral bands	Source	Scale
1946	9-Jul	black and white	USGS	1:51,000
1954	27-Sep	black and white	USGS	1:60,000
1974	16-Aug	black and white	USGS	1:80,000
1981	19-Jun	RG, Near Infrared	USGS	1:59,000
1982	5-Aug	RG, Near Infrared	USGS	1:58,000
1983	12-Sep	RG, Near Infrared	USGS	1:58,000
1984	2-Sep	RG, Near Infrared	USGS	1:58,000
1994	4-Aug	black and white	DOQQ/NAPP	1:40,000
1999	9-Sep	black and white	DOQQ/NAPP	1:40,000
2006	17-Sep	RGB	NAIP	1:12,000
2010	19-Jun	RGB, Near Infrared	NAIP	1:12,000
2013	11-Aug	RGB, Near Infrared	NAIP	1:12,000
2015	22-Jun	RGB, Near Infrared	NAIP	1:12,000
2017	30-Jun	RGB, Near Infrared	NAIP	1:12,000

8.2.2 Riparian tree cover through time

An object-based image analysis (OBIA) was performed on each photo to differentiate landscape vegetation units. OBIA separates features in an image by grouping neighboring pixels with similar characteristics including spectra, texture, size, shape, adjacency, and edge detection. OBIA is often considered superior to traditional pixel-based classification methods, as the latter only uses the digital number of each pixel to classify landscape characteristics (Blaschke, 2010). We used Trimble eCognition Software to conduct OBIA, using the multi-resolution segmentation algorithm to segment landscape features into coarse- and fine-scale objects by adjusting input settings (Appendix Table 16.1). Coarse segmentation produced larger objects and separated landscape-level units such as riparian forest, upland forest, and barren land. Fine segmentation produced smaller objects nested within the coarse segments and isolated small features such as single trees. All photo years were segmented using the same parameters, and delineated objects were exported as polygons to ArcGIS with descriptive statistics that included area and mean brightness. Coarse-scale segments were classified as riparian or non-riparian vegetation using a combination of brightness values and visual interpretation. Visual interpretation can increase classification accuracy, particularly when differences within or across photos are subtle as occurs with succession and colonization (Lillesand et al., 2015; Shalaby & Tateishi, 2007). To restrict segments to the defined potential riparian corridor, we only searched within 50 m horizontal and 3 m vertical distances from the thalweg (i.e., channel bottom), an extent identified by creating a digital elevation model in the Fluvial Corridor Toolbox (Roux et al., 2015). We then used mean brightness values, visual interpretation, and manual polygon editing to finalize coarse-scale segments. Results were comparable across photos because they had the same pixel size, radiometric resolution (i.e., number of grey levels), and were all captured during the growing season (June through September). An accuracy assessment was conducted using a binary response between riparian and non-riparian land cover types to verify the segmentation. The mean accuracy per photo was calculated to be 0.90.

We defined each year's riparian area as the sum of all polygon areas that were segmented as riparian tree cover in a photo. The 1946 and 1954 segmentations yielded riparian areas that were continuous, distinct from the uplands, and visually similar. We used the mean riparian area from these two pre-pipeline photos as the baseline riparian area for the study reaches. Each reach's riparian areas were then calculated for a given photo year and divided by its baseline area to find the proportion of riparian area present through time.

8.2.3 NDVI

Normalized Difference Vegetation Index (NDVI) is a metric used to assess vegetation greenness. Dense and healthy vegetation has high NDVI values because photosynthesizing plants absorb red light and reflect near infrared light. NDVI value differences can also vary from vegetation attributes such as physical structure, foliage type, phenology, canopy density, and homogeneity relative to photo pixel size. NDVI values are comparable across sensors and years when photos are captured during a similar time of year, such as the June 19-August 11 range included here. We performed NDVI analysis only on photos with red and near infrared bands: 1981, 2010, 2013, 2015, and 2017. Photo year 1981 was selected over the 1982–1984 photos because it more closely aligned with the seasonal timing of the 2000's photos. All photos were converted to 8-bit data scaled at 0–255 for

analysis. The 1981, 2010, 2015 and 2017 photos were captured in June and are directly comparable, while the 2013 photo was captured in August, which could cause slight seasonal differences. We used two approaches to account for potential variability across the photos, such as sun angles and shadows, that could affect calculations. First, we compared differences between the DW and R reaches *within* each photo. Second, we conducted an effect size analysis to compare riparian vegetation *across* photos. Effect size analysis calculates the difference between groups without relying on sample size to generate statistical significance (Nakagawa & Cuthill, 2007). We quantified effect size using Glass's Delta, which was the difference between DW and R NDVI means divided by the standard deviation of the Reference NDVI values (Nakagawa & Cuthill, 2007).

8.2.3.1 Tree-based assessments

We assessed the consistency between canopy conditions recorded in the field with NDVI from aerial photos. The location of each cottonwood was first collected using a GPS during 2016–2018 field surveys. A 2-meter radius buffer was created around each GPS point and spectral information was extracted and assigned to the tree. Mean NDVI was calculated for 1614 trees in the DW reach and 539 trees in the R reach. In field surveys each tree's percent live canopy was classified into groups of 0%, 1–20%, 21–40%, 41–60%, 61–80% and 81–100%, which are identified as 0%, 10%, 30%, 50%, 70%, and 90% (Section 3.3). The average NDVI value was calculated for trees in each of these six canopy classes, and the Kruskal-Wallis with Dunn statistical test was used to identify differences among classes. Additionally, we used linear models to characterize the mean NDVI per canopy class as a function of percent live canopy in each reach. Interactions models were used to determine if the model slopes differed between reaches in each year.

8.2.3.2 Focal area assessments

The Snake Creek riparian forest is affected by within-reach geomorphic complexity, so we identified ten focal areas to investigate subreach variability. Focal areas were selected to have relatively homogenous patches of riparian forest and a high proportion of cottonwoods. Focal areas were placed into four groups: high cottonwood dieback, moderate cottonwood dieback, moderate cottonwood dieback intermixed with conifers, and low cottonwood dieback (Table 8.2). Focal area boundaries were delineated using fine-scale OBIA segmentation and manually edited to exclude non-riparian vegetation such as roads. An area-weighted NDVI average was calculated from all polygons within a focal reach. We then combined areas within each of the four focal groups to identify mean NDVI values for each. The low dieback group was used as the reference. We compared the NDVI means from all other groups to this reference group, and we described the effect size using Glass's delta.

Table 8.2. The ten focal areas from NDVI analysis including group name, area, and a brief description.

Category	Name	Area (m ²)	Description
Low dieback	Ref. Upstream	5800	high percent live canopy cottonwoods
Low dieback	Ref. Downstream	7800	high percent live canopy cottonwoods
Low dieback	Ref. Squirrel	3800	high percent live canopy cottonwoods in DW reach
Moderate dieback	DW 1	5400	moderate dieback cottonwoods
Moderate dieback	DW 2	3500	moderate dieback cottonwoods
High dieback	DW 3	5900	high dieback cottonwoods
High dieback	DW 4	4500	high dieback cottonwoods
Mixed vegetation	DW 5	3600	moderate dieback cottonwoods with pinyon-juniper
Mixed vegetation	DW 6	1600	high dieback cottonwoods with fir-pinyon

8.3 Results and Discussion

8.3.1 Riparian tree cover through time

Aerial photo analysis revealed the DW and R reaches to be on different trajectories from 1946 to 2017. Tree cover in the R reach slightly expanded over the time series to finish at 106% of baseline area, going from 8.6 to 9.1 hectares (21.1 to 22.6 acres) from the 1946–1954 pre-pipeline average to 2017. In the DW reach the riparian forest declined to 55% of baseline area, going from 10.2 to 5.6 hectares (25.2 to 13.9 acres) from the pre-pipeline average to 2017 (Figure 8.2, Table 8.3). The largest decline between successive photos for the DW reach was 26% from 1954 to 1974, a period when R reach tree canopy declined by 5%. The pipeline was constructed during this period, initiating stream dewatering. An 11% increase in riparian area occurred at both the DW and R reaches from 1974 to 1981. The early 1980s were the wettest period in the flow and precipitation records (Figure 2.5 and Figure 2.7), and the high riparian area was maintained over all four photo years 1981–1984. Both periods with high frequency photos, 1981–1984 with four photos and 2010–2017 also with four photos, showed only small interannual changes.

After the wet early 1980s, a sharp decline in riparian area occurred by the next photo, in 1994. From 1984 to 1994 riparian area decreased by 14% at DW and 21% at R. The lower decline at DW could have related to its 1980s area already being relatively low. This period coincided with low precipitation and several years of lower than average runoff (Figure 2.7). The downward trend was temporary in the R reach, but it continued throughout the photo record in the DW reach (Figure 8.2). Between 1994 and 2017 the R reach riparian area averaged 109% of baseline, but in DW it was 67%. These findings were consistent with the analyses of tree growth patterns and death dates, all of which revealed declines in the DW reach riparian forest over recent decades.

Part of the identified riparian area reduction at DW was related to branch dieback and resulting fragmentation of the continuous riparian canopy. The 1946 and 1954 photos showed a dense and continuous riparian forest in both reaches. The R forest remained largely continuous throughout the

study period (Figure 8.3). However, the object-based image segmentation showed a fragmented and resultingly smaller canopy in the DW reach beginning in 1974, the first photo in the pipeline period. Figure 8.3 shows a location where ground-truthing revealed medium-high cottonwood dieback in the DW reach. Its riparian area was relatively continuous and distinct from the upland in 1946 but less so in the 1981 photo. By 2017 the forest was highly fragmented with a smaller canopy area.

Aerial photo analysis also showed upland pinyon and juniper encroachment into the riparian corridor. This invasion was more prevalent in the DW reach, where cottonwood dieback and riparian fragmentation were higher. A transition from riparian cottonwood to pinyon and juniper could result from reduced water access that favored the more drought-tolerant trees. Some of this vegetation shift was likely influenced by climate, fire suppression, and land management. However, these factors cannot explain the large discrepancy between the DW and R reaches, whereas different water management practices can.

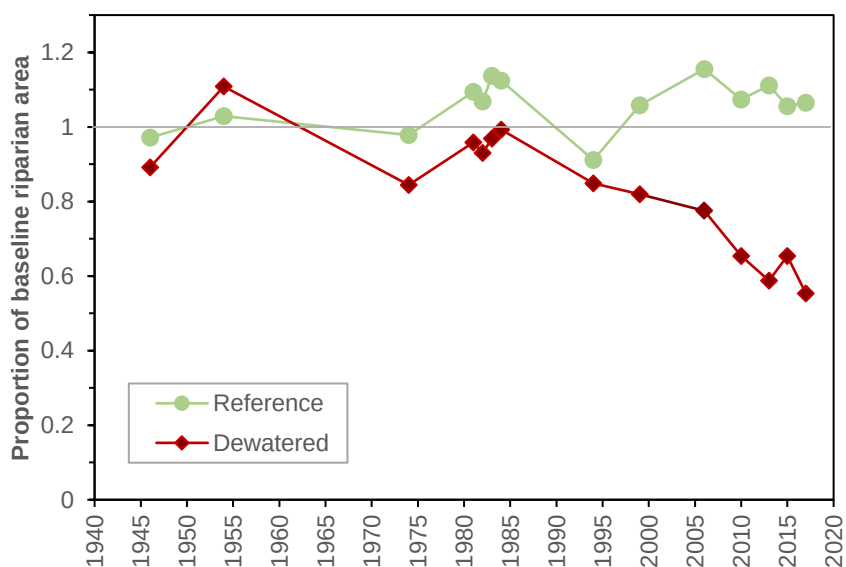


Figure 8.2. Riparian area per photo year as a proportion of baseline area in the Dewatered (dark red, diamonds) and Reference (light green, circles) reaches.

Table 8.3. Riparian area per photo year in the Dewatered and Reference reaches.

Year	Reference area (hectares/acres)	Dewatered (hectares/acres)
1946	8.3 ha (20.6 acres)	9.1 ha (22.5 acres)
1954	8.8 ha (21.8 acres)	11.3 ha (27.9 acres)
1974	8.4 ha (20.8 acres)	8.6 ha (21.3 acres)
1981	9.4 ha (23.2 acres)	9.8 ha (24.2 acres)
1982	9.2 ha (22.7 acres)	9.5 ha (23.5 acres)
1983	9.8 ha (24.1 acres)	9.9 ha (24.4 acres)
1984	9.6 ha (23.8 acres)	10.1 ha (25 acres)
1994	7.8 ha (19.3 acres)	9.7 ha (21.4 acres)
1999	9.1 ha (22.4 acres)	8.4 ha (20.7 acres)
2006	9.9 ha (24.5 acres)	7.9 ha (19.5 acres)
2010	9.2 ha (22.8 acres)	6.7 ha (16.5 acres)
2013	9.5 ha (23.6 acres)	6 ha (14.8 acres)
2015	9.1 ha (22.4 acres)	6.7 ha (16.5 acres)
2017	9.1 ha (22.6 acres)	5.6 ha (13.9 acres)

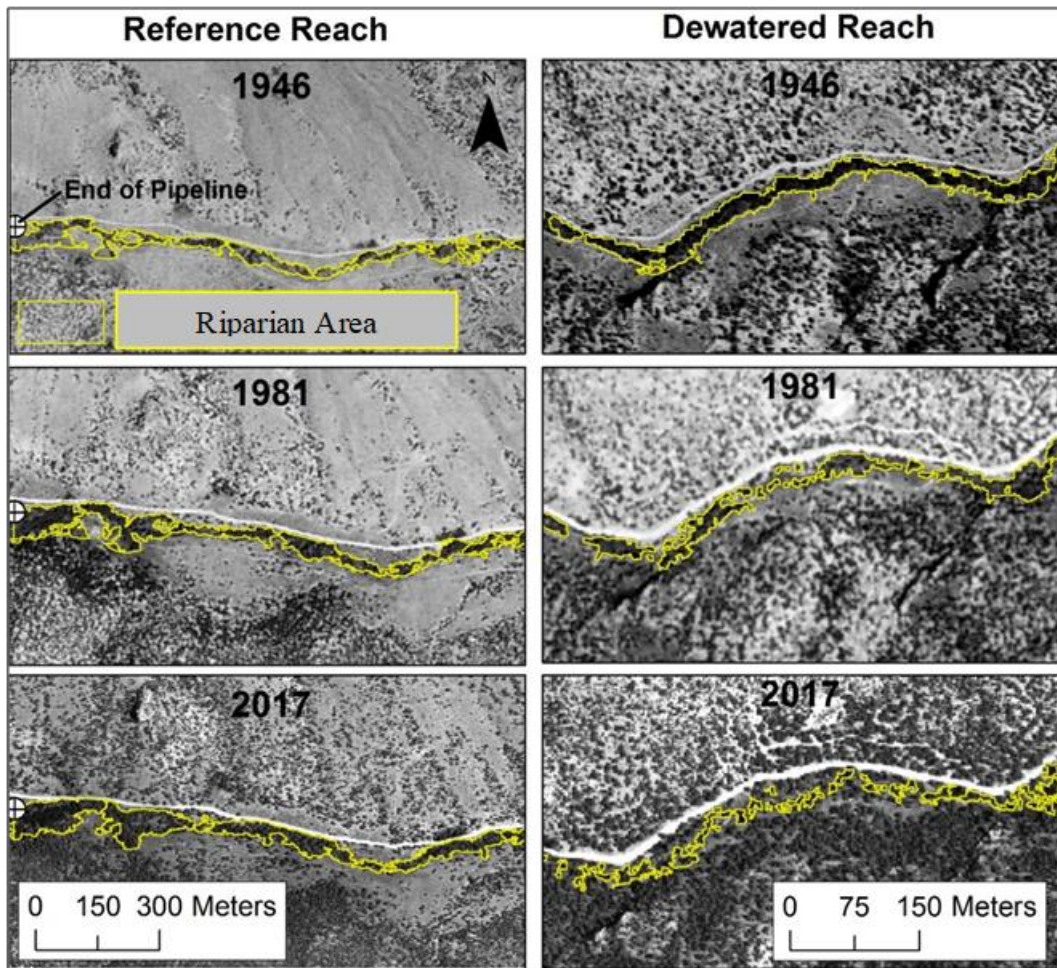


Figure 8.3. Riparian area delineations from object-based image analysis for 1946, 1981, and 2017. The Reference reach column (left) shows a riparian forest that remained intact through time. Concurrently, the DW reach (right) had riparian forest fragmentation.

8.3.2 NDVI through time

8.3.2.1 Tree-level analysis

The NDVI analysis performed on individual trees between 1981 and 2017 indicated denser vegetation in the R reach. Mean NDVI was higher in the R than DW reach in all years, and the difference between reaches increased through time (t-stats 6.54–23.91, $p < 0.001$; Figure 8.4a). The NDVI values grouped percent live canopy from the field surveys into four statistically distinct levels, with NDVI values increasing with greater percent live canopy (chi-squared = 142.1, $p < 0.001$; Figure 8.4b and Figure 8.5). The differences between reaches reflect spectral characteristics related to canopy coverage, vegetation health, and species composition.

In 1981 the mean NDVI value per tree was not significantly related to its percent live canopy identified in the 2016–2018 field surveys for either study reach ($p = 0.25$). This indicates that the canopy conditions encountered in the field surveys had not yet emerged by 1981. Our previous branch dieback and fallen dead tree analyses suggested that there were large numbers of trees with

dieback in the 1960s and 1970s. However, these trees are now lying on the ground so would be missed by this analysis of standing trees. For all subsequent NDVI analysis years, 2010–2017, the slope of regression lines linking NDVI to percent live canopy was significantly positive. This relationship was strong, with the four R^2 values for the DW reach ranging from 0.95–0.97 (Table 8.4; Appendix Table 16.2). Interaction models revealed that the slopes of regression lines were significantly higher for DW than for R in all four of these years. The higher slopes in DW indicate that dying cottonwood branches were less frequently replaced by healthy trees or other vegetation. In contrast, the lower slopes from the Reference reach and its higher average NDVI values suggest that its unhealthy trees were often surrounded or replaced by other green vegetation (Figure 8.4b).

8.3.2.2 Focal area analysis

Focal areas with low cottonwood dieback had the highest average NDVI, followed by moderate dieback, high dieback, and mixed vegetation. The mixed vegetation included cottonwoods with conifers, and it had the lowest NDVI values in all five years. This highlights that NDVI values vary by species, and forest composition can confound results so ground assessment must be considered. Glass's delta averaged 0.4 across the study period when mixed vegetation was compared to the reference low dieback areas, meaning these groups were the most dissimilar. This was the highest difference of any group, but it is generally considered only a medium-low treatment effect and no focal area results were strong.

In 1981 the low, moderate, and high dieback focal areas had nearly identical NDVI values (Figure 8.6). Thus, the focal area results support the tree-level NDVI results that indicated that riparian forest dieback among trees currently standing was minimal in 1981 compared to 2010–2017. The moderate- to low-dieback groups had very little difference, with Glass's delta averaging 0.2 from 2010–2017. For the high- and low-dieback groups the difference was slightly greater averaging 0.3 (Figure 8.6). These results were consistent with the riparian forest conditions encountered in the field. In summary, the tree-level, focal area, and reach-level NDVI analyses all suggested that the R reach has a healthier riparian forest than DW and that much of the riparian forest decline at DW occurred between 1981 and 2010.

Table 8.4. Linear models of mean NDVI value and percent live cottonwood canopy per reach. Interaction models identified if the slope of the lines was steeper in the DW reach.

Year	Reference Reach		Dewatered Reach		Interaction model p-value
	Slope (NDVI/canopy%)	R^2	Slope (NDVI/canopy%)	R^2	
1981	0.69	0.32	-0.28	0.23	0.25
2010	0.15	0.27	1.26	0.97	< 0.01
2013	0.49	0.82	1.71	0.95	< 0.001
2015	1.07	0.64	3.38	0.96	0.07
2017	1.09	0.63	3.91	0.96	0.04

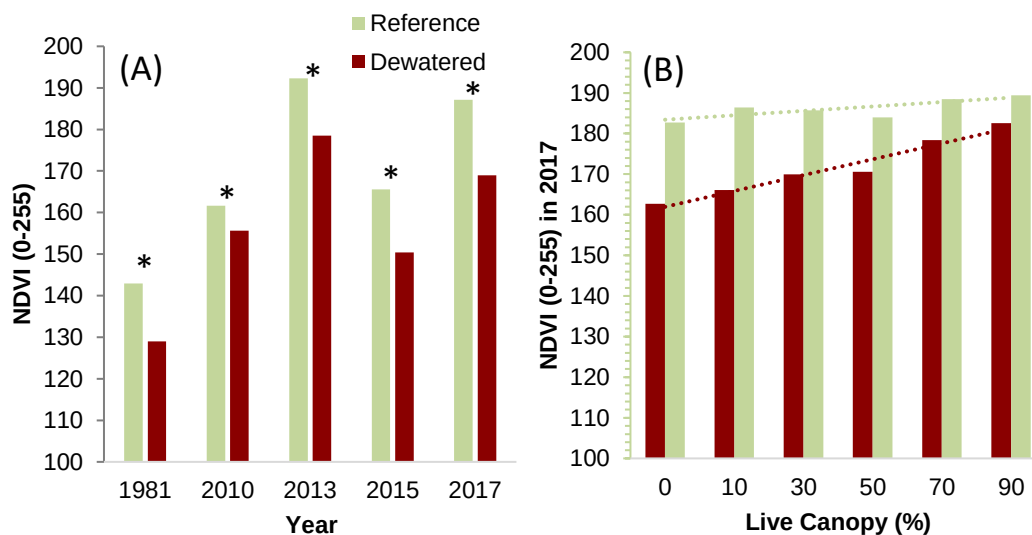


Figure 8.4. Average tree-level NDVI for Reference (light green) and Dewatered (dark red) reaches. (A) shows the average NDVI for all trees per photo, and Reference NDVI values were always significantly (*) higher. (B) shows example year 2017 NDVI values by canopy class. NDVI values increased with percent canopy and the relationship was significantly greater in DW. Note that the best fit lines displayed are for illustration, and statistical analysis was performed with a continuous x-axis (Table 8.4).

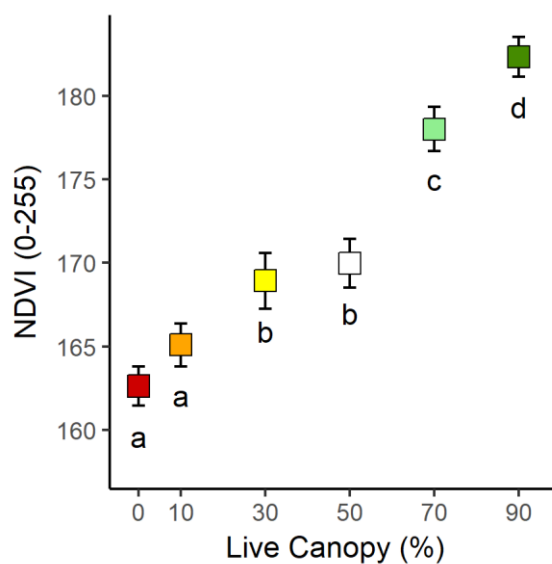


Figure 8.5. NDVI values per canopy class (mean +/- standard error) for all DW trees in 2017 (n = 1506 trees). Significantly different groups are identified with unique letters.

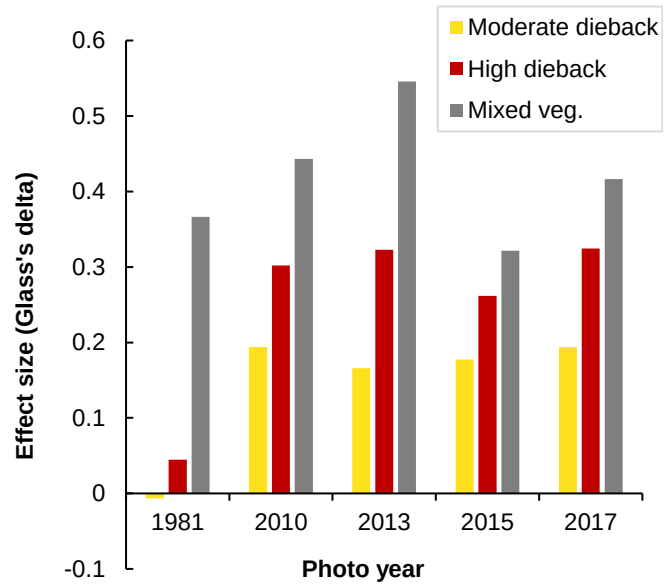


Figure 8.6. Effect sizes for NDVI means from vegetation groups in the focal area analysis. All effect sizes were in comparison to the low dieback group that was used as a reference.

9. Stand-Level Cottonwood Production

9.1 Introduction

Analyzing individual cottonwood trees provides information at the tree level, but upscaling growth to the stand level can provide a broader view of the Snake Creek riparian forest. Forest decline may not be detected in investigations of tree rings from living trees alone. Stand-level characterization can help account for complications caused by trees responding to changes in the local growing environment. For example, competitive dynamics change when a tree's neighbors die from drought, but its ring width could *increase* if shading and root-competition had previously limited its growth. Stand-level forest productivity is commonly characterized using tree basal area production (Chamagne et al., 2017; Hogg et al., 2008). Forest production can decline as trees die or the forest perimeter retracts, independent of tree-level production. Comparing stand-level production across stream reaches requires diverse datasets, such as those available from our previous analyses.

9.2 Methods

We reconstructed cottonwood forest production through time using two complementary approaches that combined tree ring width (Section 4) and either aerial photos or cross-dated trees. Forest production was expressed as annual stand-level basal area increment (SBAI), which was the cumulative basal area increment (BAI, Section 4.2.1) of all cottonwoods in the stand. We quantified annual forest production by stream length ($\text{m}^2\text{km}^{-1}\text{yr}^{-1}$) instead of on a per area basis, which is commonly used for upland forests instead of linear features like rivers, because assigning a width to the riparian corridor would have constrained production measurements independent of hydrogeomorphic conditions.

The first approach used aerial photos and combined tree-level BAI, tree density, and photo-delineated riparian areas to calculate SBAI for the DW, RU, and RD reaches. RU and RD were then merged into a Reference reach for consistency with methods in the aerial photo analysis (Section 8.2.1). Aerial photos showed riparian area in 14 years between 1946 and 2017 (Table 8.1), and we linearly interpolated annual riparian areas for all years between photos. Cottonwood density was calculated using the DW reach in 2017 as the standard. The live cottonwood count from field survey ($n = 1132$) was divided into riparian area in the 2017 photo. We found one cottonwood for every 49.86 m^2 of riparian area in the DW reach, a density also used for the adjacent RU and RD reaches. This conservative procedure reduced differences between reaches, and it was necessary because large portions of the RU riparian forest were composed of other tree species and the aerial photo RD reach extended beyond the area surveyed in field study (Figure 8.1). Customized cottonwood densities for RU and RD would have affected their absolute values of forest production but would not have changed trends through time. We calculated annual SBAI values by multiplying each reach's BAI by its riparian area, then dividing by area per tree.

In the second approach we calculated SBAI by multiplying each reach's BAI by its number of living cottonwoods per year, 1940–2016. We first counted the living trees ($>20 \text{ cm}$ diameter) during field survey. Establishment dates from the subset of cross-dated trees were used to calculate the number of trees alive in each year since 1940. Next, *standing* dead trees were surveyed and their death dates

were identified from cross-dating a random sample that permitted calculation of the proportion of the dead trees that had been alive in every year. *Fallen* dead trees were surveyed with death dates identified for those in the DW reach. Fallen dead were not surveyed at RU or RD, so they were assumed to match the proportional abundance of standing-to-fallen dead trees found in the DW reach (49.8% standing, 50.2% fallen). This meant that for RU and RD we added a nearly equal number of fallen dead to match the standing dead. They were conservatively assumed to have the same death date distribution as found in the DW reach. Finally, the number of live trees per year was multiplied by each reach's average annual BAI and divided by the reach length that was surveyed and had cottonwoods (4.38 km in DW, 0.31 km in RU, 0.30 km in RD) to yield a final SBAI values for each reach.

9.3 Results

9.3.1 Aerial Photo approach

Snake Creek's cottonwood forest grew at 0.36 and $0.24 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$ in the DW and Reference reaches in the 1946–1961 baseline period (Figure 9.1). Post-pipeline growth averaged 67% and 97% of those values in DW and Reference. In the post-pipeline period the two SBAI trajectories differed substantially, declining by $31 \text{ cm}^2\text{km}^{-1}\text{yr}^{-1}$ (1.5% per year) in the DW reach and $0.04 \text{ cm}^2\text{km}^{-1}\text{yr}^{-1}$ (0.016%) in the Reference reach. A large SBAI decline occurred in the DW reach from around 2000 through 2016, corresponding to a divergence in riparian areas between study reaches. Standardizing each reach to its pre-pipeline baseline growth rate shows that differences between reaches increased in the 1960s and again in the 2000s as both BAI and riparian area declined in the DW reach (Figure 9.1b).

DW reach SBAI peaked in 1954 ($0.52 \text{ m}^2\text{km}^{-1}$) and again in 1983–1984 ($0.47 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$). A similar 1983–1984 peak ($0.42 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$) occurred in Reference reach trees (Figure 9.1). The lowest production occurred in DW from 2013–2016 ($0.07 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$), while the lowest 3-year period in the Reference reach was much higher at $0.16 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$ from 1996–1998.

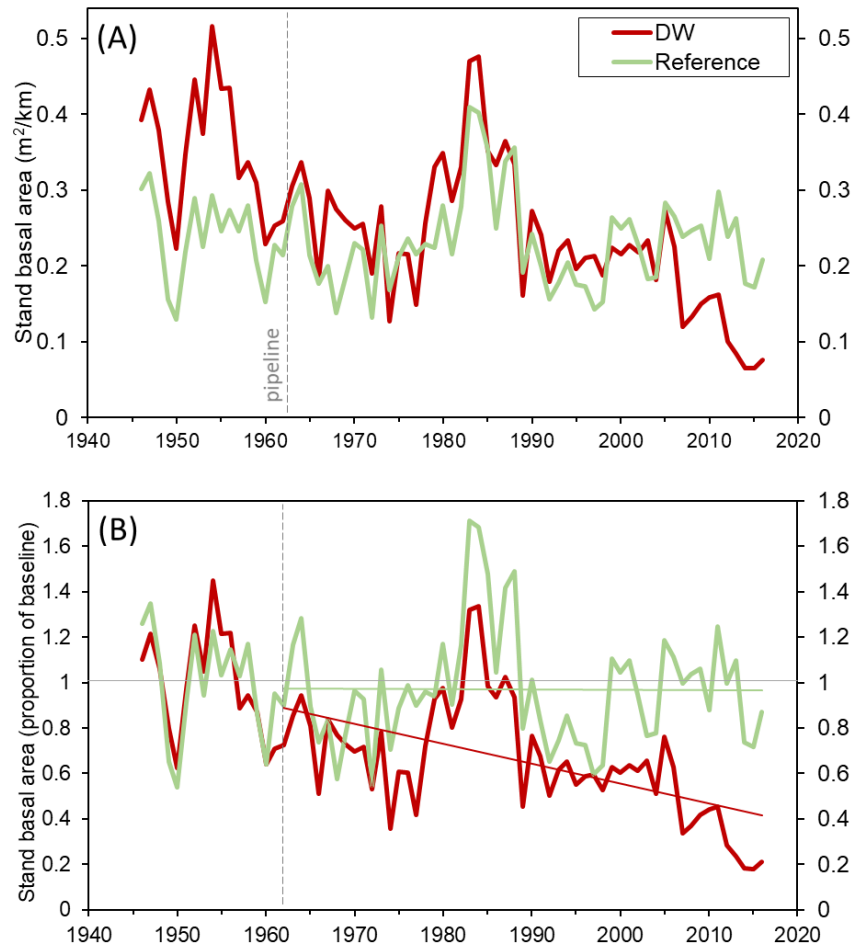


Figure 9.1. (A) Stand basal area increment (SBAI) per river kilometer for the DW and Reference reaches using the air photo approach. (B) shows SBAI values relative to the baseline period before pipeline installation, with 1962–2016 growth for DW and Reference averaging 67% and 97% of pre-pipeline average while declining at rates of 31 and $0.4 \text{ cm}^2\text{km}^{-1}\text{yr}^{-1}$ (straight lines).

9.3.2 Tree-count approach

Over the entire 1940–2016 study period the DW reach’s riparian forest SBAI averaged $0.29 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$, while RU and RD were higher at 0.48 and $0.38 \text{ m}^2\text{km}^{-1}\text{yr}^{-1}$. Higher production in reference reaches was partially caused by their higher cottonwood densities. Comparisons to pre-pipeline growth more effectively isolated the influence of water access. The RD reach had 110% of its pre-pipeline growth in the post-pipeline period and had no long-term trend. In contrast, the DW and RU reaches both declined through time and averaged 65% and 69% of pre-pipeline growth in the post-pipeline period (Figure 9.2b). By 2017, 44% of DW and 22% of RU trees had died which led to significant differences between the reaches by the end of the study period. The DW reach’s higher mortality rate was partially offset by its younger trees that established from 1940s–1970s, lowering its early years reach-level production compared to RU where all trees established prior to 1940. Differences between DW and RU emerged in the mid-2000s (Figure 9.2), which was also the

beginning of the period with the highest mortality in the DW reach. The last decade, 2007–2016, had a growth rate of 31% the pre-pipeline rate at DW, 52% at RU, and 95% at RD.

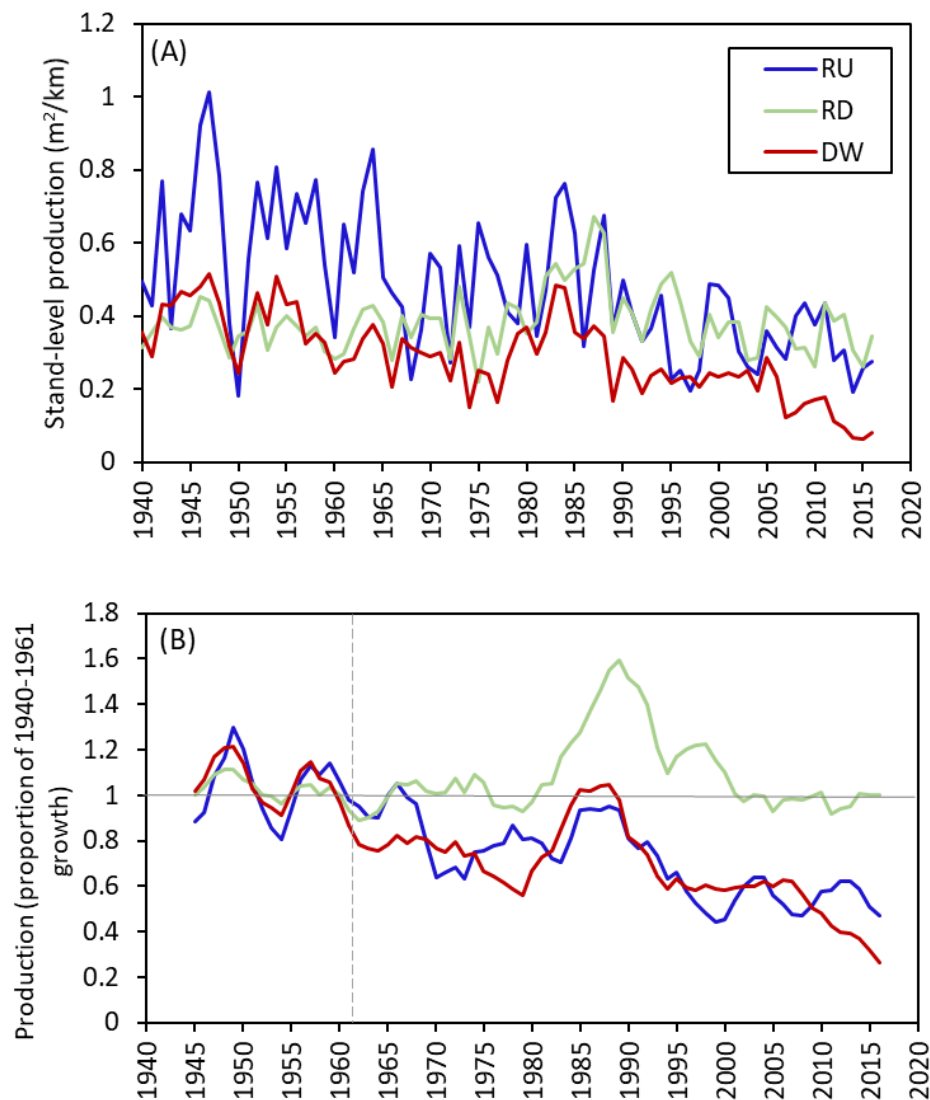


Figure 9.2. Stand-level basal area increment (SBAI (m^2/km)) for cottonwoods by reach, shown as (A) annual values or (B) a 5-year running mean scaled as a proportion of the pre-pipeline baseline growth (horizontal gray line). Note that the vertical dashed-gray line in 1961 represents pipeline installation.

9.4 Discussion

Scaling cottonwood growth from individual trees to stands characterizes the Snake Creek riparian forest as a landscape-scale habitat. Of the two approaches used, neither showed a threshold change in forest production after pipeline installation in 1961. Instead, there was a gradual decline in DW reach riparian forest production from 1940 to 2016. We did not find a similar decline in RD or the combined Reference reach, although in the tree-level analysis RU trees exhibited a similar decline

until the DW drop-off in the 2000s. The declines at DW and RU were likely caused by different processes, however. The RU decline could not have been caused by stream dewatering because the reach is upstream of the pipeline. The type of decline observed would be expected if competition limited tree ring growth, a phenomenon most likely in the RU reach because of its high cottonwood density and the tall dense evergreens. This conclusion of non-hydrologic controls being most influential in the RU reach is consistent with our isotope analysis and analysis of hydroclimatic drivers controlling tree ring growth, where we found that flow and precipitation were least correlated with cottonwood growth in the RU reach (Figure 6.1). Given the more open forest canopy in DW, water availability remains a more likely cause of its SBAI decline even though a threshold change following pipeline installation did not occur in the BAI and SBAI analyses as in isotope analysis where DW trees $\delta^{13}\text{C}$ values increased immediately after pipeline installation.

The two complementary research approaches led to similar stand-level conclusions that forest production declined in the DW reach over the last half-century. Each approach used cottonwood BAI and inclusion of aerial photos or tree-counts overcame some limitations while retaining others. The assumption of equal cottonwood density by reach in aerial photo analysis affected absolute production values in the Reference reach, but this should not have affected trends or the main conclusion that significant differences occurred between reaches. The tree-count approach was affected by the inevitability that dead trees rot, disappear, and eventually avoid sample inclusion. This problem increases further back in time, but the rates of rot should have been similar across reaches. Together the approaches were deemed robust and informative, and their convergence lends strength to the conclusion that DW's riparian forest production has declined since stream dewatering.

10. Groundwater Monitoring

10.1 Introduction

Riparian groundwater is typically connected to stream stage and is essential for many riparian plants (Naiman et al., 2010). The rooting depth of phreatophytes such as cottonwoods is related to the prevailing groundwater depth during plant maturation (Butler et al., 2007; Rood et al., 2011). Decreases in water table depth can result in tree dieback or death (Cooper et al., 2003; Grismer, 2018; Shafroth et al., 2000). In arid regions, streamflow commonly recharges riparian groundwater, and when streamflow is removed the water table can decline below the plant rooting zone. The hydrologic and ecological consequences of a water table decline are influenced by valley setting, lithology, and the gradient between surface water and groundwater. Because of this complexity, the most direct way to characterize groundwater available to Snake Creek's riparian cottonwoods is to measure the elevation of surface water and groundwater using stream staff gauges and groundwater monitoring wells. These measurements provide information needed to calculate hydrologic gradients and determine groundwater-surface water relationships. Additionally, borehole cuttings extracted during well drilling provide information on subsurface sediment characteristics, including soil development, alluvium depth, grain size, rock type, and moisture content.

10.2 Methods

We installed eight shallow groundwater monitoring wells between July 25 and 28, 2018 in locations selected for proximity to cottonwoods, access from the road, and avoidance of known cultural artifacts. Wells 1–7 were in the dewatered (DW) reach, and Well 8 was in the reference downstream (RD) reach. Six of the well locations overlapped with electrical resistivity survey locations (Section 11), which improved interpretation of both datasets.

Wells were drilled to 3.8 to 6.1 m below ground using a sonic drill (Model DSC550-18, *Sonic Drill Company*). The boreholes were 15 cm in diameter, and the native soil and rock extracted from them was scattered for disposal. Sonic drilling directs high-frequency resonant energy into the subsurface where it reduces porosity and causes liquefaction of soil and rock. This process enabled the drill to advance through cobbles, boulders, and bedrock. Sonic drilling was also selected to minimize landscape disturbance, and rubber mats were placed on vegetation during construction to limit disturbance from the track-mounted drill rig.

The groundwater monitoring wells were installed along five transects perpendicular to Snake Creek. In two areas one well was installed, and in three areas two wells were installed 7–13 meters apart. Wells were constructed using 3.8 cm machine slotted PVC in the lower 3.05 m (Wells 1–7) or lower 1.52 m (Well 8) to allow groundwater movement in and out of the well. A solid PVC riser extended above the slotted pipe to within 10 cm of the ground surface. The boreholes were backfilled around the well casing with coarse sand extending to 0.6 m above the slotted section then sealed above with bentonite chips. Wells were finished flush to the ground surface with a cemented vault.

Each well transect was paired with a staff gauge installed in the channel (Figure 10.1). Staff gauges were constructed of perforated 3.8 cm PVC tube secured by bolting to a boulder (Staff 4), bolting to a streamside tree (Staffs 2 and 5), or tying to a t-post that was driven into the stream bed downstream

of a protective debris jam (Staffs 1 and 3). Every well and staff gauge in the DW reach ($n = 11$) was dry at the time of installation, while the well and staff gauge in the RD reach were in water.

Well Transect 1 is 40 m below the pipeline inlet near the top of the DW reach (Figure 10.2). Well Transect 2 is located downstream and 100 m upstream of Squirrel Springs. Cottonwood distribution widened 60 m below the transect, likely indicating increased near-surface groundwater access close to Squirrel Springs. Well Transect 3 is 800 m downstream from Squirrel Springs. Well Transect 4 is 500 m further downstream, and still 1000 m upstream of the pipeline outlet. Well Transect 5 is 120 m below the pipeline outlet in the RD reach. The channel curved just upstream of Transects 4 and 5, and it is likely that some groundwater measured in the wells came from upstream, so relationships between groundwater and surface water elevations were likely complicated by upstream delivery on these transects.

Water levels were monitored from July 2018 to October 2019. The Baker Creek #3 Snow Course (NRCS) recorded April 1 snow water equivalent (SWE) to be 82% of the post-1942 average in 2018 and 141% of average in 2019. After 1 April 2019, however, SWE continued to rise until May 30 at the new Wheeler Peak snonet station (NRCS), indicating that 2019 had >141% of average snowpack and should generate much higher than average runoff.

A RuggedTROLL 100 pressure transducer (In-Situ, Inc.) was suspended near the bottom of each well or staff gauge to record hourly water levels from 1 August 2018 through 4 September 2019. The continuously logging pressure transducers provided information on seasonal relationship of surface water-groundwater exchange and on water in the cottonwood root zone.

Topography was surveyed at all five transects using a Topcon laser level to characterize gradients and exchanges between the surface water and riparian groundwater (Figure 10.1). Each cross-section included the top of the well vaults and can be re-established by placing a tape over the wells and extending it across the channel where a yellow-capped rebar was installed to protrude 5 cm above ground. Staff gauges 2–5 were not directly on their transects because hydraulically protected locations were uncommon. However, all staff gages were within 5 m of the cross-section and channel bed elevation offsets were measured to translate relative water levels.

The borehole cuttings extracted during drilling provided information on subsurface conditions. We documented changes in color and texture from the cuttings. 24 samples were collected in sealed bags for laboratory analysis at Colorado State University. We identified each sample's color using a Munsell color book and determined particle size by hand. The Pole Canyon limestone has historically been assumed to drive natural groundwater recharge, so we investigated carbonate prevalence in the borehole cuttings using a 3% hydrochloric acid (HCl) solution. Gravimetric water content was determined by weighing samples, oven drying them at 55° C for 48 hours, and reweighing them. Some samples were not collected and sealed immediately upon extraction, which introduced noise into water content measurements.

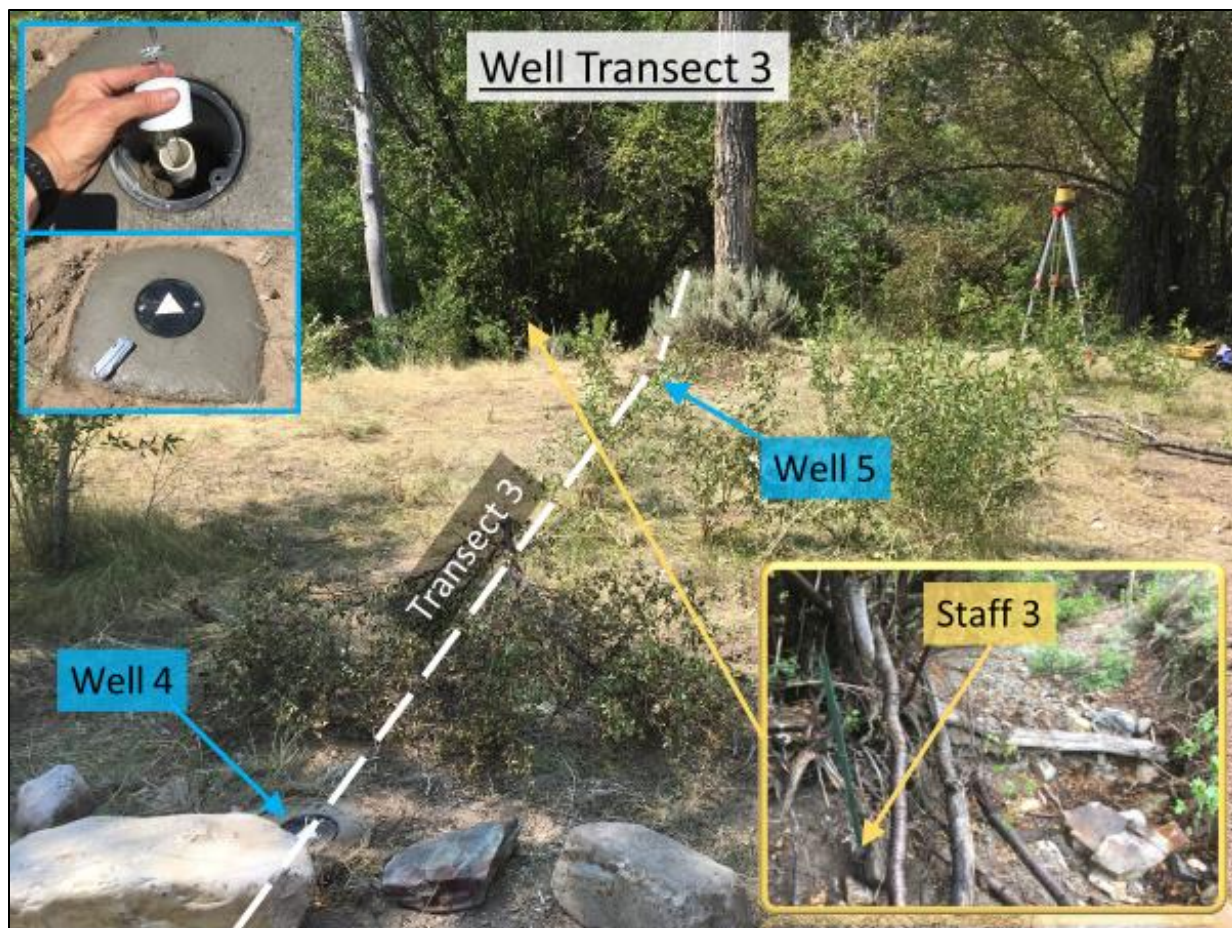


Figure 10.1. Groundwater well Transect 3 as an example. Wells 4 and 5 (top-left inset) were installed in the floodplain at foreground, and Staff 3 (bottom-right inset) was affixed to a t-post in the channel.



Figure 10.2. Locations of well and staff gauge transects along Snake Creek. Transects 1–4 are in the DW reach, and Transect 5 is in the RD reach. The DW reach extended from pipeline inlet (left) to outlet (right). See Table 10.1 for the wells and staff gauges on the transects.

10.3 Results

10.3.1 Water levels

Groundwater and surface water levels varied within and among monitoring transects (Table 10.1). Water levels at Transect 1 revealed that the Snake Creek channel was dry for most of the year just below the pipeline inlet, where 2019 snowmelt runoff produced streamflow from April 28–30 and from May 13 to August 22 (Figure 10.3). The 6.1 m deep Well 1 was dry throughout the study period, indicating either an unsaturated zone beneath the stream channel or a steeply losing hydrologic gradient (i.e., surface water to groundwater directionality) that exceeded the 39% slope from the channel bed to the well bottom. This transect clearly indicated that groundwater was deep and below the cottonwood root zone except for the trees growing very close to the channel when there was flow.

Transect 2 was located 100 m upstream from Squirrel Springs. Its two wells and staff gauge indicated strongly losing hydrology and groundwater within 5 m of the ground surface only when supported by surface flow (Figure 10.4). The channel was dry from the time of installation until snowmelt runoff began on June 6. Groundwater was detected in Well 3 (5.3 m from channel) one hour after instream flow began, and it was in Well 2 (17.2 m from channel) after three hours. These time lags were likely caused by water gradually extending from the channel. A seasonal runoff peak occurred on June 7, when there was a 19% downward (i.e., losing) water table slope from the channel to Well 3 and a 15% slope to Well 2. Well 2 was completed to 5.3 m below ground and only had water from June 6 to June 24, indicating that high runoff was needed to support groundwater within 5.3 m of the ground surface at this location. By August 1, the water table gradient from the channel to Well 3 steepened to 39% as streamflow and groundwater declined. Surface flow ended on August 22, and the groundwater level in Well 3 dropped sharply one hour later, declining 80 cm to below the bottom of the well within 16 hours of surface flow cessation.

Transect 3 was the only transect in the DW reach to have groundwater even when surface flow was absent. The channel and wells were dry when installed in July 2018, but on 12 October 2018 water was detected in Well 5 (4.2 m from channel), and it was found in Well 4 (12.2 m from channel) five days later (Figure 10.5). Water was only detectable in Wells 4 and 5 when it was within 5.2 m and 4.8 m of ground surface. Groundwater rose to 3.8 m below ground at Well 5 within three days during the October rise, but it took 70 days to rise to a similar level at Well 4. Groundwater remained around this depth through February 2019, when it gradually rose 0.5 m more by April. Snake Creek surface flow began on April 21, and groundwater rose to about the same elevation as surface water. This water table rise began the same hour in the channel and wells, indicating a strongly connected surface and subsurface hydrologic system. Water levels rose and fell together throughout the runoff season, and surface water levels being higher indicated a slightly losing hydrologic gradient. On July 29 the staff gauge's pressure transducer indicated that surface water had stopped flowing. However, field reconnaissance on October 2 found that a trickle of flow had remained all runoff season, indicating that the pressure transducer was not in the deepest part of the channel. Because surface water persisted, we could not evaluate if groundwater immediately and strongly declines upon channel drying like at Transect 2.

Transect 4, the most downstream in the DW reach, was dry until spring runoff began on April 22 (Figure 10.6). Water was detected first in the channel, then one hour later in Well 6 (17.9 m from channel) and three hours later in Well 7 (9.8 m from channel). The unexpected order of water detection in the wells could have been influenced by three factors. First, variable hydraulic conductivity and preferential flow paths could have accelerated the surface water-groundwater connection at Well 6. Second, the water table could have started closer to the bottom of Well 6 and required less rise to be detected. Third, the sinuous Snake Creek channel curved just upstream of the transect, allowing groundwater levels to be influenced by upstream seepage not strictly perpendicular connections to the channel along the transect. Water in the wells was higher than surface water on the transect throughout the runoff period, but the channel sinuosity obscured interpretation of surface water-groundwater gradients. Like Transect 2, groundwater elevation declined relative to surface water throughout the runoff season. This indicated a hydrologic gradient that either became less gaining or, more likely, more losing through time. As with Transect 3, both surface and groundwater persisted through the end of study, precluding evaluation of groundwater response to surface water drying.

Transect 5 was in the RD reach where groundwater and surface water persisted for the entire study period. From August 2018 to February 2019, groundwater in Well 8 averaged 44 cm below the channel bottom, but snowmelt runoff began on 23 April 2019, caused the water table to rise, and reversed the surface water-groundwater gradient (Figure 10.7). Groundwater increased by an average of 107 cm during May through August and was 63 cm above the channel bottom. The corresponding surface water rise was 10 cm to 23 cm above the channel bottom between these periods. The cross-section indicates that the channel was hydrologically losing in the first period and gaining in the second period. However, as with Transect 4, channel sinuosity may have affected water levels here and complicated interpretation of the hydrologic gradient.

The five transects were used to interpret longitudinal (i.e., downstream) characteristics of Snake Creek hydrology. Surface water runoff in 2019 was detected first at the three transects below Squirrel Springs in the upstream-to-downstream direction, appearing at Transects 3, 4, and 5 on April 21, 22, and 23 (Figure 10.8). The largest snowmelt runoff spike progressed in a downstream direction as well. This rise first occurred at Transect 1 on June 5, then at Transects 2 and 3 on June 6, and at Transects 4 and 5 on June 7. The magnitude of the surface water rises generally decreased downstream, indicating flashier conditions upstream and more hydrologically stable conditions downstream. This is the opposite of what generally occurs in mountain streams, but it was affected by subsurface geology and the pipeline. Snake Creek was very strongly losing at the top of the DW reach and still strongly losing as it approached Squirrel Springs. At both DW reach transects below Squirrel Springs the hydrologic gradient was more neutral. Finally, in the RD reach Snake Creek appeared to fully shift to a gaining stream during the runoff season, but this conclusion may partially be an artifact caused by down-valley flow of groundwater through a bend in the creek.

Table 10.1. Groundwater wells and surface water staff gauges with the number of days with measured water and the maximum water level between 1 August 2018 and 4 September 2019.

Transect	Instrument	Total Depth below ground (m)	Days with measurable water (out of 400 days)	Max water level above ground surface (m)
1	Well 1	6.1	0	< -6.1
1	Staff 1	0	104	0.60
2	Well 2	5.3	18	-2.52
2	Well 3	5.2	79	-0.73
2	Staff 2	0	78	0.36
3	Well 4	4.8	323	0.55
3	Well 5	4.9	328	0.33
3	Staff 3	0	101 ¹	0.54
4	Well 6	5.5	136	0.80
4	Well 7	5.6	136	0.70
4	Staff 4	0	136	0.54
5	Well 8	3.8	400	0.90
5	Staff 5	0	400	0.41

¹ note that the staff gauge was installed too high to detect the trickle of water persisting from July 30 to September 4, 2019. If detected, that would make 136 days and match the transect's wells.

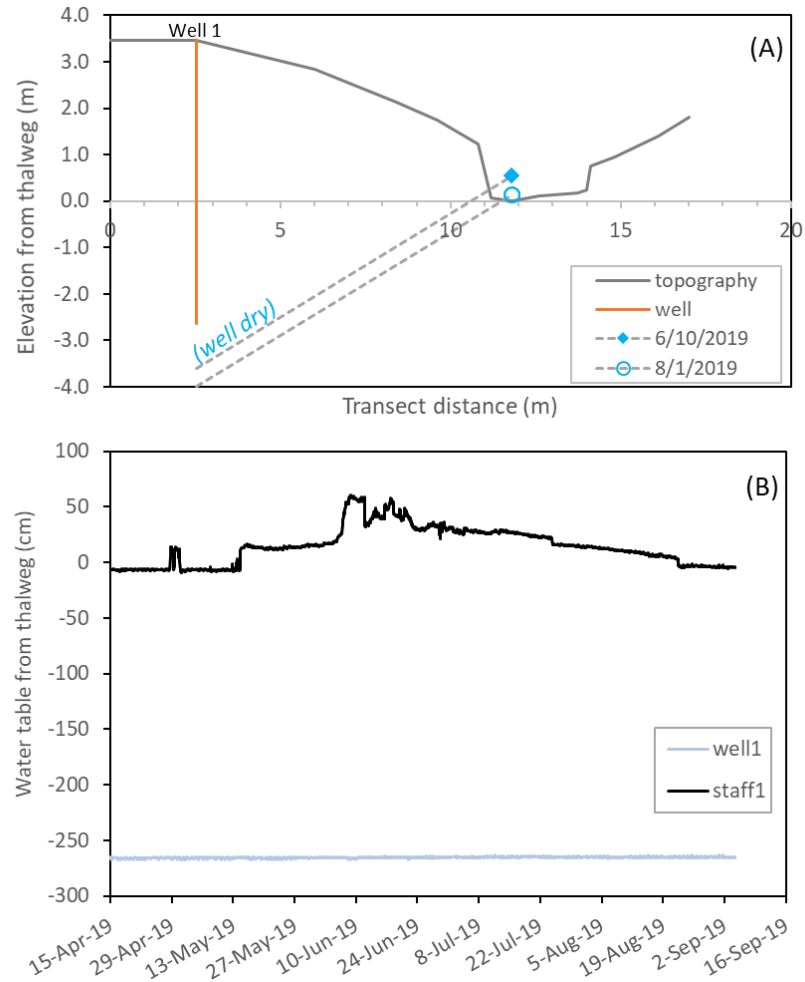


Figure 10.3. (A) Cross-section at Transect 1 and (B) time-series of water table depths at Well 1 and Staff gauge 1 from April to September 2019. Dashed lines in (A) represent that the water table was always below Well 1 and the alluvium under the channel was either very strongly losing or unsaturated. In (B) water depths are shown relative to the bottom of the channel. Note that flat portions in the lines occur when the well or staff gauge was dry (Well 1 was always dry).

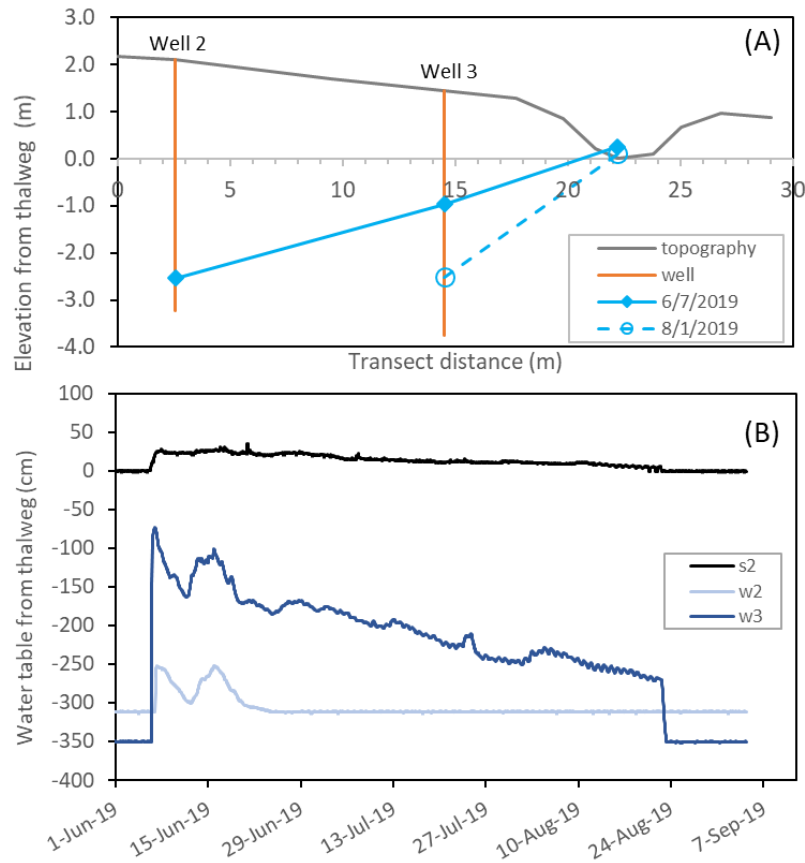


Figure 10.4. (A) Cross-section at Transect 2 and (B) time-series of water table depths at Well 2, Well 3, and Staff gauge 2 from June to September 2019. Blue lines in (A) show the water table on the seasonal peak on 7 Jun 2019 and later in the runoff season on 1 Aug 2019. In (B) water depths are shown relative to the bottom of the channel. Note that flat portions in the lines occur when the well or staff gauge was dry.

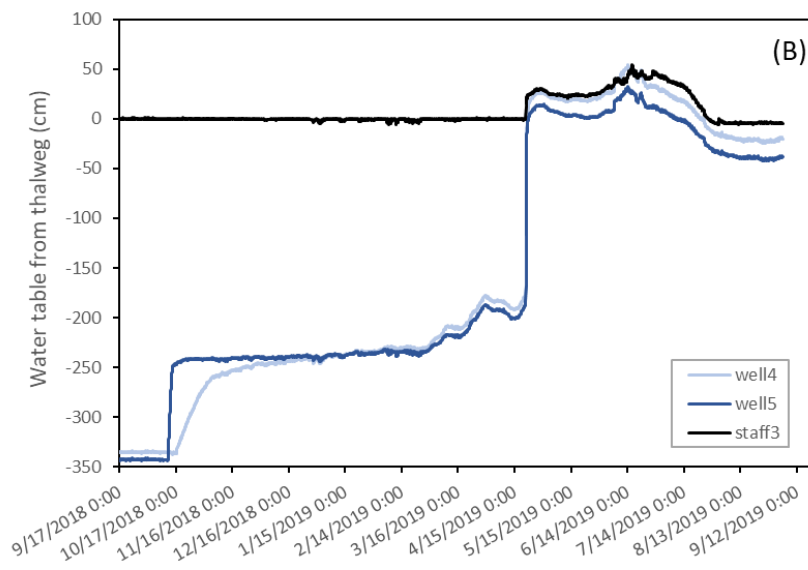
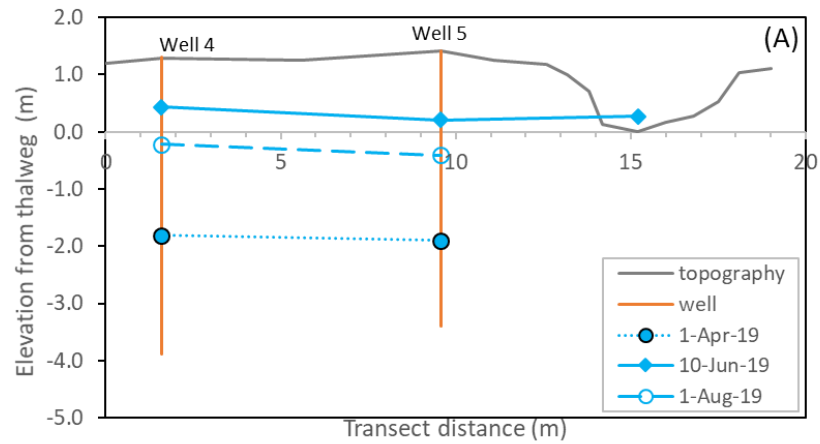


Figure 10.5. (A) Cross-section at Transect 3 and (B) time-series of water table depths at Well 4, Well 5, and Staff gauge 3 from September 2018 to September 2019. Blue lines in (A) show the water table before, during, and after peak seasonal runoff. In (B) water depths are shown relative to the bottom of the channel. Note that flat portions in the lines occur when the well or staff gauge was dry.

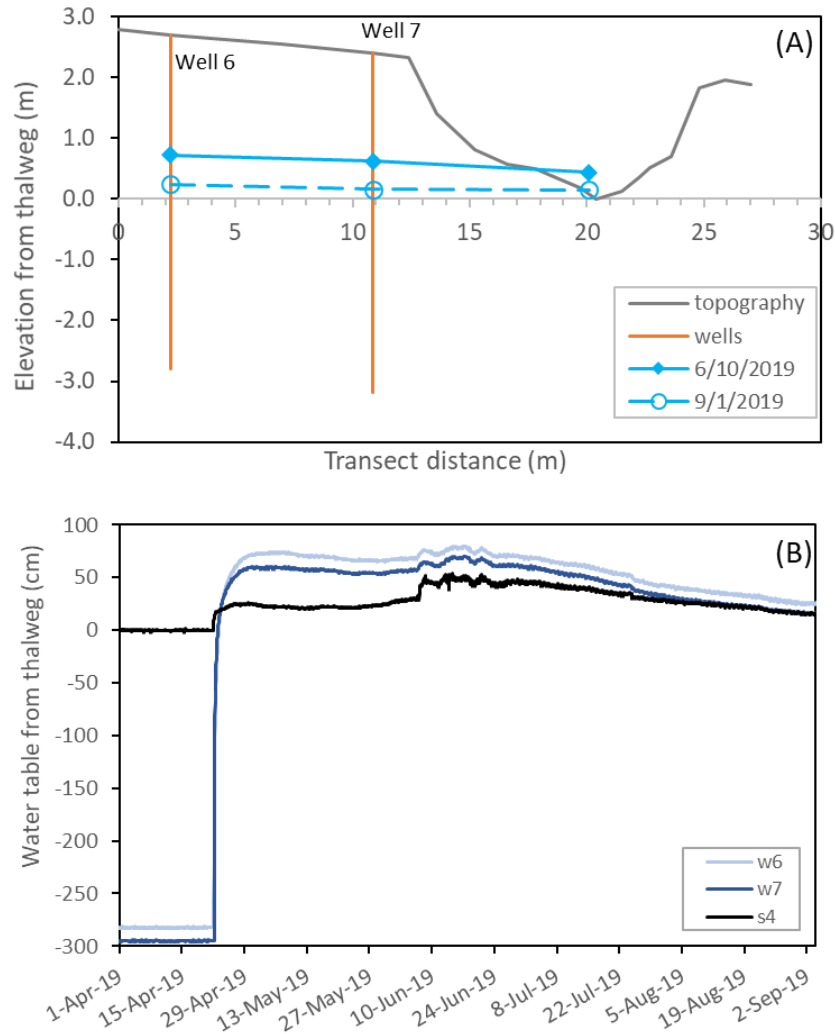


Figure 10.6. (A) Cross-section at Transect 4 and (B) time-series of water table depths at Well 6, Well 7, and Staff gauge 4 from April to September 2019. Blue lines in (A) show the water table during and after peak runoff, and note the modern channel is highly incised from the former floodplain. In (B) water depths are shown relative to the bottom of the channel. Note that flat portions in the lines occur when the well or staff gauge was dry.

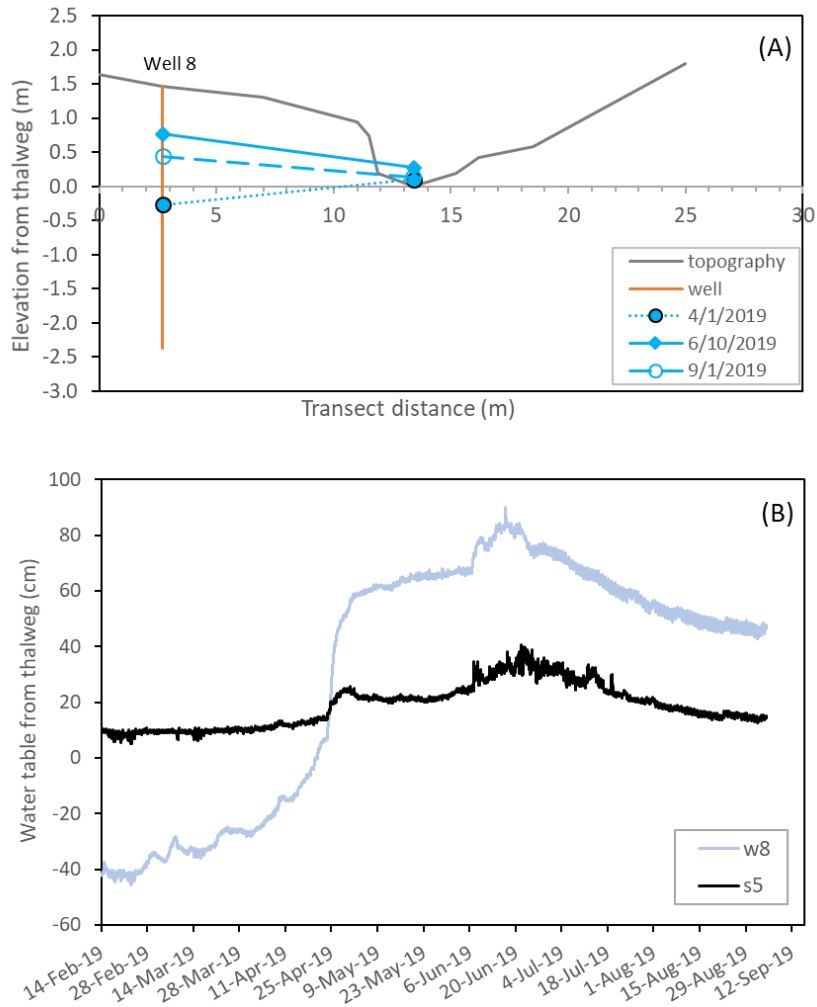


Figure 10.7. (A) Cross-section at Transect 5 and (B) time-series of water table depths at Well 4, Well 5, and Staff gauge 3 from February to September 2019. Blue lines in (A) show the water table before, during, and after peak seasonal runoff. In (B) water depths are shown relative to the bottom of the channel.

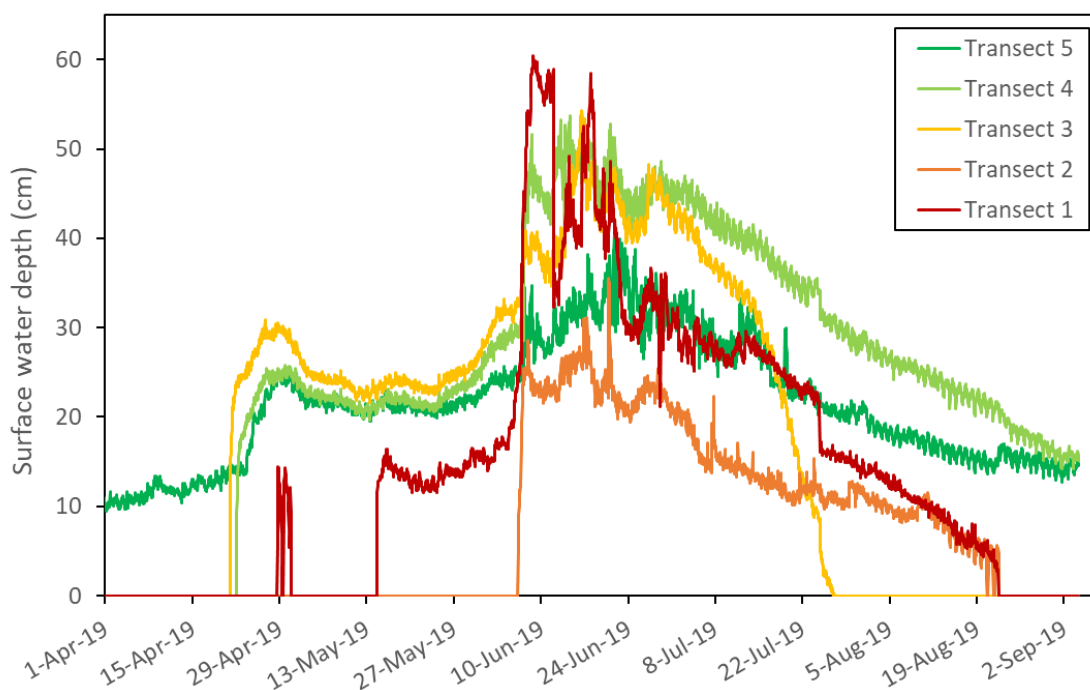


Figure 10.8. Surface water levels at the five monitoring transects from April to September 2019. Transects 1 to 4 progress downstream in the DW reach, and Transect 5 is in the RD reach.

10.3.2 Well borehole extractions

Borehole sediment samples had very low water content throughout the likely cottonwood rooting zone (Table 10.2). Samples collected from the DW reach had a median gravimetric water content of 1.4% and mean of 2.9%. The three samples from Well 8, located in the RD reach where water was present, had a median gravimetric water content of 16.9% and mean of 15.5%. The DW values are suspected to be below the wilting point for cottonwoods, although delayed sample collection likely lowered some values. Some samples had very low water content because they were mostly comprised of solid or pulverized rock, not *in-situ* soil with pore space capable of supporting soil moisture and plant roots.

The eight monitoring wells extended through a total of 41 m of Snake Creek valley bottom. Samples collected included unconsolidated upland soil, geochemically reduced wetland soil, sand, gravel, cobble, and portions of broken boulders and bedrock. Nineteen of 24 (79%) samples reacted with HCl, indicating carbonate-derived minerals associated with limestone or dolomite (Table 10.2). The presence of highly fractured and porous carbonate bedrock has been widely assumed to be the cause of surface water recharge to the groundwater system in the DW reach, and our results support its prevalence. 71% of samples were in the 2.5Y color palette. The most common soil textures were loamy sand (29%), sandy loam (29%), and loam (25%). However, many sample textures deviated from their *in-situ* form because the sonic drilling pulverized rocks and reduced grain sizes.

Table 10.2. Summary of the extracted material from well boreholes.

Sample Name ¹	Gravimetric water content	Acid Test Reaction ²	Color	Texture	Pulverized Rock?	Estimated % of sample that is rock
MW1-1-6	0.011	strong	2.5Y 4/2	loamy sand	n	40
MW1-2-12	0.002	strong	2.5Y 4/2	gravel	n	80
MW1-3-20	0.005	strong	2.5Y 4/3	loamy sand	y	50
MW2-1-2	0.014	strong	2.5Y 3/3	sandy loam	n	50
MW2-2-5.7	0.007	strong	2.5Y 3/3	sandy loam	n	40
MW3-1-3	0.030	strong	2.5Y 2.5/1	loam	n	10
MW3-2-8	0.000	strong	5YR 7/2	loamy sand	y	90
MW3-3-16	0.032	no	10YR 4/4	loamy sand	n	80
MW4-1-5	0.116	strong	2.5Y 3/2	loam	n	0
MW4-2-8	0.024	no	2.5Y 3/2	silt loam	n	70
MW4-3-13	0.002	strong	2.5Y 4/2	sandy loam	y	80
MW4-4-17	0.072	no	10YR 5/4	sandy loam	n	30
MW5-1-9	0.026	no	2.5Y 3/2	loamy sand	n	60
MW5-2-12	0.001	strong & weak	2.5Y 3/2	sandy loam	y	95
MW5-3-16	0.077	no	2.5Y 4/3	sandy loam	n	10
MW6-1-8	0.055	strong	10YR 3/2	silt loam	n	0
MW6-2-15	0.009	strong	2.5Y 4/2	sandy loam	y	50
MW6-3-20	0.010	strong & no	2.5Y 4/2	loamy sand	y	70
MW7-1-3	0.072	strong	10YR 3/1	loam	n	0
MW7-2-10	0.047	weak	2.5Y 3/2	loam	n	0
MW7-3-15	0.002	strong & no	NA	rocks	y	95
MW8-1-3	0.169	strong	10YR 4/2	loam	n	0
MW8-2-8	0.219	strong	2.5Y 4/2	loam	n	0
MW8-3-12	0.077	weak	2.5Y 5/3	loamy sand	n	20

¹ Nomenclature is “MW” for monitoring well, then numbers correspond to the well number, sample number from that well, and sample collection depth (ft)

² Multiple reactions are reported if the sample clearly contained multiple rock types

10.4 Discussion

Groundwater and surface water measurements indicated that riparian groundwater in Snake Creek’s DW reach was supported by surface water. In 2019 some surface flow entered the DW reach when it

exceeded the intake capacity of the pipeline, and other surface flow emerged from groundwater discharge areas at and below Squirrel Springs. Squirrel Springs was a clear inflection point in surface and subsurface hydrologic processes in the DW reach, and it was likely driven by hydrologically restrictive lithology and groundwater springs. While groundwater wells showed strongly losing gradients in the upper half of the DW reach, below Squirrel Springs groundwater occurred independent of surface water, and surface water-groundwater gradients were nearly neutral. This indicated that surface water loss to the groundwater system was small below Squirrel Springs. Directly below Squirrel Springs, the riparian corridor is within the footprint of former Bonita Mine mining activities and is likely heavily altered from natural conditions, including clearing of riparian trees. Where cottonwoods first appear downstream of the mine, they have unusually high canopy survival and high tree ring growth compared to other trees in the DW reach.

The water table was > 4.8 m below ground in all DW reach groundwater wells upon installation in late July 2018, and it remained deep until 2019 snowmelt runoff began in April–June. Groundwater this deep was below the presumed cottonwood rooting zone (Beechie et al., 2006; Holloway et al., 2017; Rood et al., 2011; Singer et al., 2013) and would not be capable of supporting phreatophytes in this arid environment. Cottonwoods in the Great Basin region require water in excess of local precipitation, such as from near-surface riparian groundwater. Much of the cottonwood decline documented in this report is likely due to infrequent runoff occurring in the DW reach due to pipeline extractions. Water levels in Snake Creek and its riparian corridor varied greatly within and between years because of snowpack, recent precipitation, and seasonal runoff patterns. For example, no water occurred in the DW reach in August 2018, while in 2019 surface water and groundwater both persisted in multiple areas of the reach. Continued Snake Creek water level monitoring will further increase our understanding of groundwater-surface water interactions in this complicated hydrologic system.

11. Near-Surface Geophysics

11.1 Introduction

Shallow geophysical techniques can be used to characterize complex subsurface lithologic and hydrologic conditions such as faults, fractures, bedrock voids, and potential groundwater flow pathways. Electrical resistivity is one such technique widely used because it can cover large areas with non-destructive and repeatable sampling (Johnson et al., 2012). Electrical resistivity is an intrinsic measure of a material's ability to conduct an electrical current. Conductance, or its inverse resistance, of a geologic material is related to the following ranked factors: moisture content, porosity and permeability, pore fluid dissolved solids, temperature, and mineralogy. Electrical resistivity has been used in mountainous environments to characterize hydrologic connections between streams and adjacent groundwater (Covino et al., 2011; Harman et al., 2016; Sparacino et al., 2019). We used electrical resistivity at Snake Creek to 1) characterize the complex lithology related to different bedrock units and alluvial deposits, and 2) better understand how alluvial groundwater was connected to the Snake Creek channel and riparian corridor.

The current report details results from only five of the ten electrical resistivity study areas, and the full results are presented in a report the NPS Water Resources Division delivered to GRBA on January 30, 2019.

11.2 Methods

Electrical resistivity surveys were conducted at Snake Creek on May 17–19, 2018, when all Snake Creek flow was being diverted into the pipeline. We used an AGI SuperSting™ Wi-Fi R4 electrical resistivity meter and passive electrode cabling. The system introduced a current into a sequential series of transmitting electrodes then read the resulting voltage between two receiving electrodes. Measured current and voltage were converted to apparent resistivity. Greater survey line length and electrode spacing increased the depth of investigation, but resolution decreased with electrode spacing. Resolution is roughly one-half of electrode spacing, so a line with 1 m electrode spacing should be able to resolve features with dimensions greater than 0.5 m. General patterns of lithological changes and depth to groundwater were discernable. The transition between unsaturated and saturated subsurface conditions, generally assumed to be the water table, was also generally identifiable if present within the depth of investigation of a given survey line.

Ten survey lines were established in the Snake Creek study area. Three lines were longitudinal and parallel to Snake Creek on the alluvial terrace. Seven lines were cross-sections at flowing or diverted locations of Snake Creek. Line placement was selected to investigate areas and features of interest related to the hydrogeology or riparian conditions within the study area. After a line was set and contact resistance was minimized, the resistivity meter collected data for approximately 45–60 minutes. GPS readings were taken at the start and end of each electrode cable length and other selected electrode locations for mapping and terrain correction. The surveys included areas above, within, and below the DW reach. Lines were placed at an electrode spacing of between 1 and 3 meters, resulting in a range of depth of investigation between 11 m and 39 m (Table 11.1). Approximate line locations on the geologic cross-section are shown on Figure 11.1.

Table 11.1. Summary of Snake Creek electrical resistivity survey lines. Lines 1, 2, 5, 6 and 7 are described in the following results.

Transect Line No.	Orientation	Electrode spacing (m)	Max depth (m)	RMS error (%)	L2 error
1	Longitudinal	2	26.1	4.96	0.98
2	X-section	1	14.0	5.89	0.96
3	X-section	1.5	11.7	4.54	0.83
4	X-section	1	14.2	3.26	0.43
5	X-section	1	13.3	4.22	0.71
6	Longitudinal	3	32.7	5.16	0.74
7	X-section	1	17.6	6.64	0.90
8	X-section	1	12.0	6.72	0.70
9	X-section	1	11.1	4.31	0.52
10	Longitudinal	3	39.2	3.82	0.58

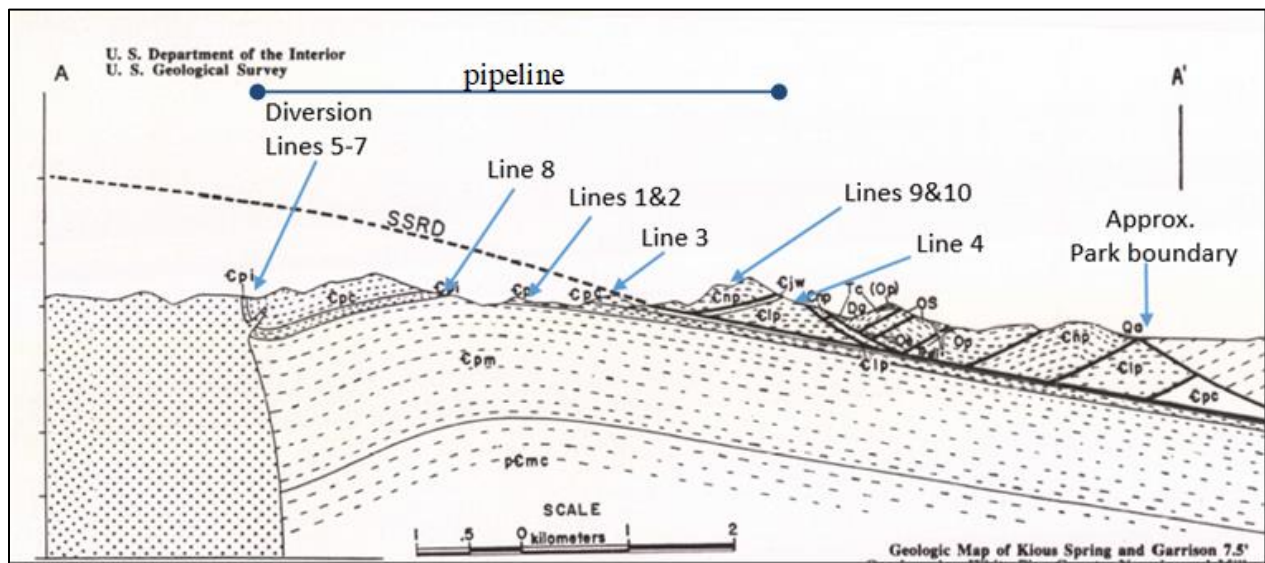


Figure 11.1. Geologic cross-section showing the approximate location of electrical resistivity survey lines and other features in the Snake Creek study area. The pipeline inlet was at “Lines 5–7” and the outlet was immediately above Line 4. Geologic cross-section modified from (McGrew & Miller, 1995).

Raw data returns from the survey lines were imported into the processing software EarthImager2D™. Inversion was used to model the data by reconstructing the subsurface resistivity distribution from measured voltage and current data, enabling interpretation of subsurface geologic and hydrologic conditions. Filtering options to remove noisy data and to fit the measured and predicted results were used, and then a final inversion was run. Errors are reported as the root mean

squared (RMS) and L2 errors. The L2 error represents a goodness of fit that is weighted to the number of data points, and it is more robust than RMS. All error values were considered acceptable and are shown in Table 11.1. Color scaling is unique to each inversion profile to maximize the range of resistivity values measured, so similar colors on different profiles represent different resistivity values and interpreted results. Elevation data at electrode points were used to correct for change in ground surface elevation.

Groundwater monitoring well construction allowed for the results of the electrical resistivity surveys to be compared against observed lithology and level of saturation. Well data helped corroborate findings of the geophysical surveys.

11.3 Results and Discussion

11.3.1 Transect line 5

The first three electrical resistivity transect discussed were near the pipeline inlet. Transect 5 was a cross-section across a perennially flowing portion of Snake Creek 10 m upstream from the inlet. The electrical resistivity results indicated that bedrock of the Pole Canyon limestone was shallow on the south side of the creek, which was consistent with observations of exposed bedrock beyond the southern end of the survey line (Figure 11.2). A thin interval of low resistivity soil mantled the underlying bedrock, which exhibited an undulating and likely weathered surface. High resistivity indicated the bedrock was unsaturated. The subsurface became saturated closer to the channel, and imaging indicated hydrologically losing conditions below the creek. Thus, surface water was being lost to the subsurface upstream of the pipeline inlet. The north end of the survey line was underlain by a thicker section of alluvium, likely a result of a valley widening here and the presence of an ephemeral wash entering Snake Creek.

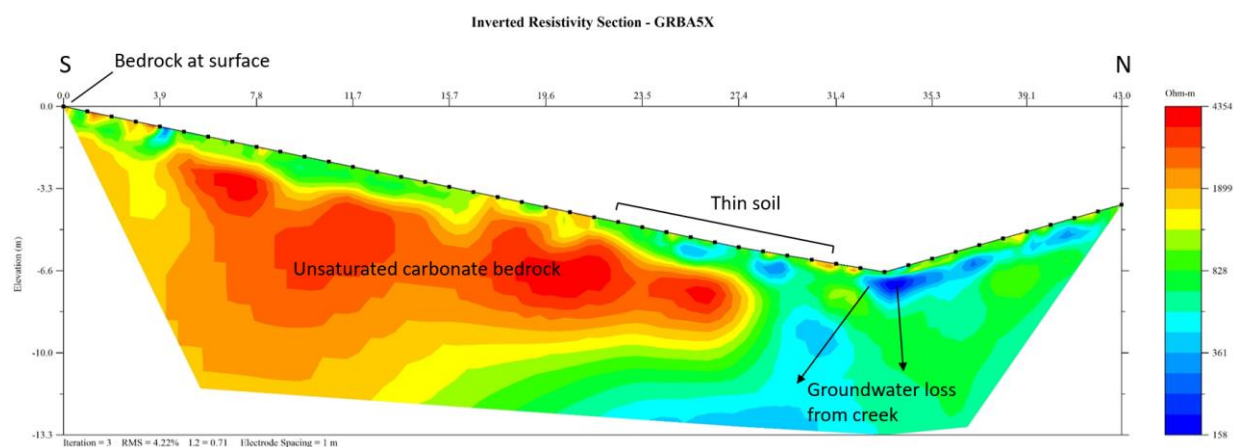


Figure 11.2. Upstream view of electrical resistivity profile at Transect 5.

11.3.2 Transect line 6

ER Transect 6 was a longitudinal profile beginning just upstream of Transect 5 and extending 15 m to the pipeline inlet before continuing 140 m farther into the DW reach. The wide electrode spacing enabled inference to approximately 33 m depth, much deeper than intersecting cross-sections at

Transects 5 and 7. This survey line was placed on the alluvial terrace above Snake Creek because of the poor conditions for electrodes on the streambed. The upstream end of the profile indicated near-surface saturation and water table, consistent with Transect 5 and as expected adjacent to the flowing creek (Figure 11.3). However, immediately downstream of the pipeline inlet the saturated zone rapidly lowered. The depth to saturation continued to drop farther downstream until appearing to flatten approximately 16–17 m below ground. This supports observations that high and prolonged streamflow exceeding the capacity of the pipeline are needed to sustain continuous flow in the DW reach, as the large volume of unsaturated alluvium could absorb substantial inflow.

Snake Creek monitoring Well 1 was constructed along Transect 6 on July 27, 2018 (Figure 11.3). Extractions from the borehole material support our interpretation of the electrical resistivity results, with 1.4 m of organic loamy soil followed by gravel and fine sand (green in inversion profile) overlying a very coarse limestone cobble and boulder alluvium to 6.1 meters deep (yellow and orange in the inversion profile). The well drilling encountered no groundwater, even though Snake Creek did flow past the closed pipeline inlet one day before because of fish management activities. The inversion profile suggests that Well 1 was still several meters above the water table on May 18, 2018. This steep groundwater gradient below the pipeline inlet suggests that the alluvial water table needs surface flow to maintain rooting-zone saturation.

We observed an apparently recently formed sink in the Snake Creek channel approximately 40 m downstream of the pipeline inlet (Figure 11.4). This feature aligned with an anomaly in the inversion profile that could represent a karst feature such as a collapse structure in the limestone bedrock.

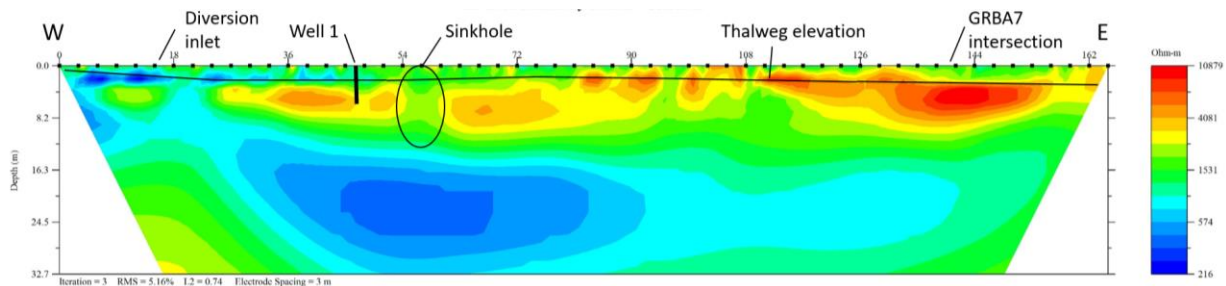


Figure 11.3. Electrical resistivity inversion profile at Transect 6. Black line represents approximate depth of Snake Creek below the terrace where the line was conducted. Downstream is to the right. The vertical line is monitoring Well #1, which never contained water, and “GRBA7” is electrical resistivity Transect 7.



Figure 11.4. Researcher standing in a sinkhole below the Snake Creek pipeline inlet structure, part of which is visible in the background.

11.3.3 Transect line 7

Transect 7 was a cross-section perpendicular to Transect 6 that was located 125 m below the pipeline inlet in the DW reach. It had a similar bedrock condition and inversion profile as Transect 5 in the perennially-flowing reach. The water table appeared to be deep, either below the maximum depth of resolution (8 m below the channel) or at the bottom of the profile 6–7 m below the channel (Figure 11.5). In contrast to the other DW reach cross-sections further downstream, there was no evidence of lateral groundwater flow into Snake Creek.

A high resistivity anomaly on the southern end of the inversion profile could represent dry Pole Canyon limestone bedrock containing an air-filled void, but because confidence in results decreases towards the margins of the inversion we are unsure (Figure 11.5). Two high resistance features on the north side of the channel could be large limestone boulders in the alluvium, or they could be anomalies caused by the unusually noisy nature of this transect.

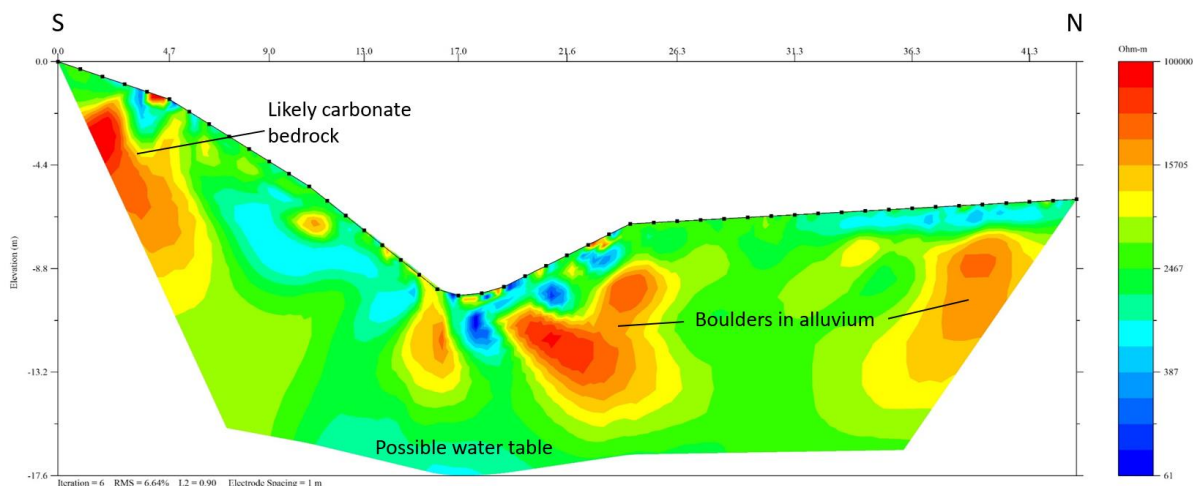


Figure 11.5. Upstream view of the electrical resistivity inversion profile at Transect 7, with annotations interpreting the profile.

11.3.4 Transect lines 1 and 2

Electrical resistivity Transects 1 and 2 were conducted as a longitudinal/cross-section pair just upstream of the Squirrel Springs camping area. The longitudinal Transect 1 showed relatively flat bedding with approximately 1.5 m of fine-grained soil above coarser channel alluvium (Figure 11.6a). The apparent water table depth was 6.5 m below ground, or approximately 5 m below the channel. This saturated zone was shallowest near the channel, which revealed losing conditions. Transect 2 showed some indication that groundwater entered the valley from the south (Figure 11.6b), but any groundwater did not generate surface flow. Additionally, neither of the 5.2 and 5.3 m deep monitoring wells encountered groundwater when they were constructed in July 2018, two months after electrical resistivity surveys.

Comparing inversion profiles from Transects 1 and 2 to the borehole material from nearby Well 2 and Well 3, there was general agreement but resolution was higher in well logs. Both inversion profiles and the well logs indicated a surficial soil and fine-grained alluvium overlaying a coarser cobble and boulder dominated unsaturated zone. The boreholes penetrated boulders larger than the 15 cm borehole diameter, resulting in rock “cores” in the returns. These were initially interpreted to be bedrock, but deeper drilling encountered more alluvium.

Squirrel Springs discharges from the dolomite on the north side of Snake Creek approximately 100 m downstream from the Transect 1 – Transect 2 intersection. This lateral groundwater input to Snake Creek is likely coupled with additional groundwater inputs that are either too deep or too small to emerge as surface water. The difference between the Transect 2 cross-section, which had losing groundwater conditions, and the lateral inflow area near Squirrel Springs is striking when viewed as a map of cottonwood distribution (Figure 11.7). In the losing reach cottonwoods were only located immediately adjacent to Snake Creek. Downstream, where lateral groundwater input occurred, the width of the groundwater-dependent cottonwood forest greatly expanded.

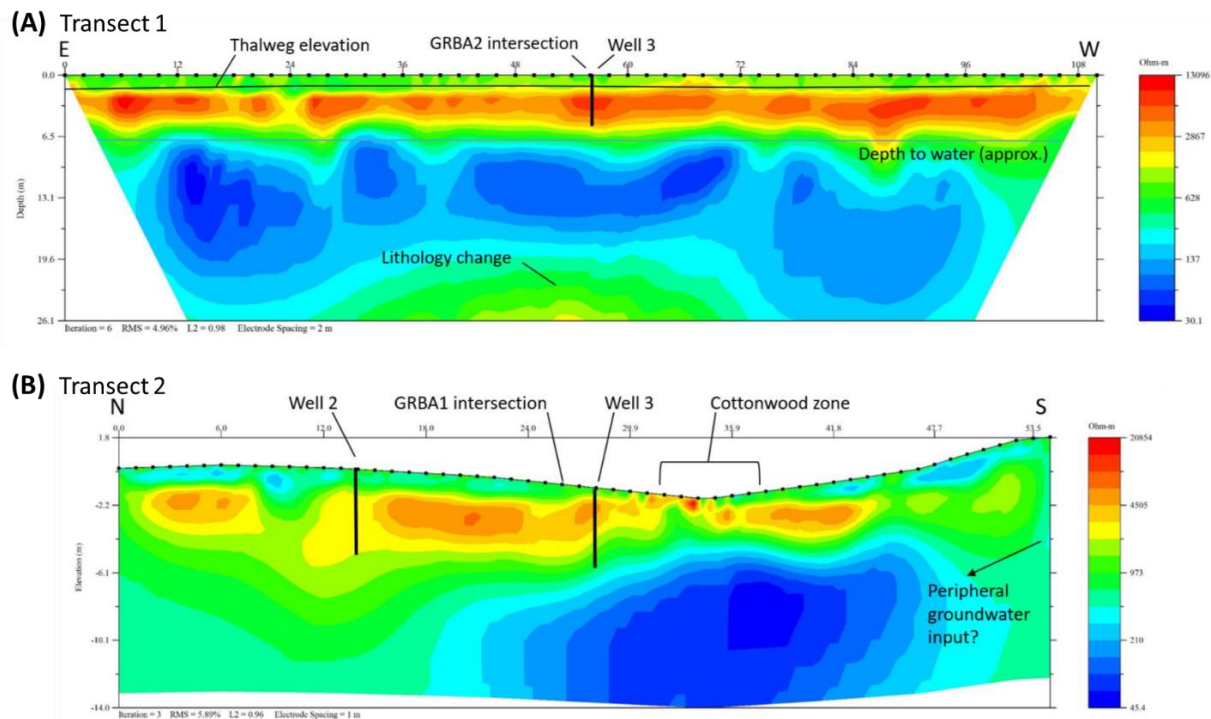


Figure 11.6. Electrical resistivity inversion profiles for (A) longitudinal Transect 1 and (B) valley cross-section Transect 2. Colors show resistivities, and annotations describe the location of wells, channel thalweg, the other resistivity profile (GRBA1 and GRBA2 for Transects 1 and 2), and profile interpretations. Note that x- and y-axis scales differ. See Figure 11.7 for transect orientation.

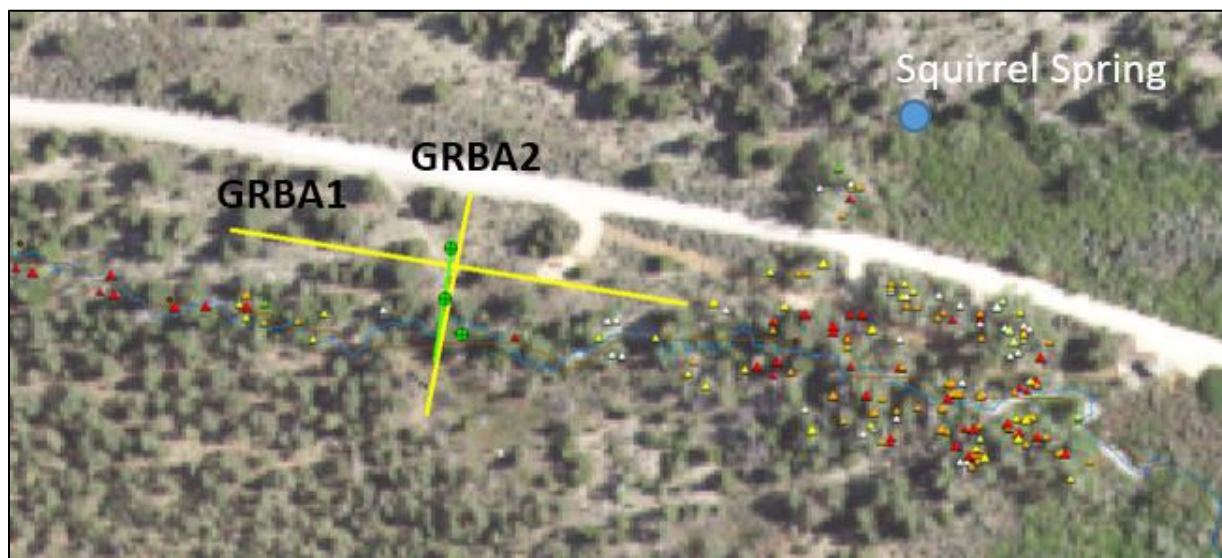


Figure 11.7. Distribution of cottonwoods (triangles) upstream (left) and downstream of lateral groundwater inputs at Squirrel Springs. Trees were narrowly constrained along the channel in the upstream losing reach, and the cottonwood zone expanded where near-surface groundwater occurred. Trees are colored by percent live canopy (red = 0%, orange = 10%, yellow = 30%, white = 50%, light green = 70%, green = 90%). Yellow lines are electrical resistivity transects 1 and 2. Green circles, from top to bottom, are Well 2, Well 3, and Staff Gauge 2 on groundwater Transect 2.

11.3.5 Implications and correspondence to groundwater wells

Electrical resistivity profiles indicated a strong loss of surface water into the channel alluvium in the DW reach. There was evidence that Snake Creek is losing above the pipeline inlet as well, but the reach's perennial flow resupplied the slow losses and maintained near surface saturation that was accessible to vegetation. Karst development in carbonate bedrock, the presence of faults and fractures, and the Snake Creek pipeline likely all contribute to surface water loss.

Electrical resistivity results also indicated the importance of lateral groundwater inputs from carbonate and quartzite units adjacent to Snake Creek. Most of these inputs likely occur in the subsurface where they do not produce surface flow but can reach riparian plants. The Snake Creek channel below Squirrel Springs likely experienced such inflow, although electrical resistivity surveying was not conducted there to confirm this.

Water table depth across the DW reach was generally interpreted to be deeper than that required to sustain cottonwoods. If the conditions encountered during field sampling persist, the riparian vegetation is unlikely to survive. The monitoring wells were constructed to a depth of 6 m to capture the cottonwood rooting zone. No wells in the DW reach encountered groundwater at the time of installation, supporting interpretations from electrical resistivity surveys. Although the resolution between datasets varied, general agreement between monitoring well and electrical resistivity investigations supported that our interpretations of the electrical resistivity data accurately portrayed subsurface conditions at Snake Creek.

12. Channel and valley morphology

12.1 Introduction

Stream channel dimensions adjust in response to the water and sediment inputs that are delivered over time (Mackin, 1948). Changes to either input can disrupt the channel's dynamic equilibrium and cause channel adjustments that include incision, aggradation, and changes in channel dimensions. Channels commonly incise in human-impacted watersheds when sediment loads decrease, flows increase, or banks destabilize after riparian vegetation is lost (David et al., 2009; Liébault et al., 2002). Water management actions can destabilize channels by modifying water and sediment inputs and by changing the water available to riparian vegetation. Impacts from water diversions on mountain rivers and streams can manifest as reductions in bed material size (Baker et al., 2011), cross-sectional area (McKay & King, 2006), channel width from plant encroachment (Ryan, 1997), and diversity of riparian vegetation (Caskey et al., 2015). Water diversions can also cause channel enlargement if riparian vegetation is killed (Erskine et al., 2012). We conducted cross-sectional surveys along Snake Creek to investigate if the geomorphic character of the channel was affected by the artificial dewatering. Although portions of the DW reach were visibly incised, the vertical, lateral, and longitudinal extents of incision were unknown.

The topography and geomorphology of mountain valleys naturally influence river channel shape and size, and channel dimensions respond to changes in valley width and slope (Lane, 1954; Montgomery & Buffington, 1997). Steeper channels have greater unit stream power and shear stress, conditions that tend to enlarge channels by increasing bank erosion and sediment transport capacity. Additionally, narrower valleys have less capacity for energy dissipation through overbank flows, so their concentrated flows can also create a larger channel. Snake Creek crosses complex geology within the study reaches. Variations in channel slope and valley confinement could be potential causes of larger channel dimensions in the DW reach, independent of differences in flow. We investigated the relationships between these two variables in an attempt to investigate a natural cause capable of explaining any changes in channel dimensions across study reaches.

12.2 Methods

12.2.1 Study reaches

In August 2017 geomorphic measurements were taken to compare the DW reach with upstream and downstream reference reaches (RU and RD, respectively). The DW reach was the 4900 m portion of Snake Creek that has experienced stream dewatering (Figure 12.1). We expanded the RU and RD reaches from cottonwood analyses to increase sample size and channel length. The RU reach was 870 m long and located immediately upstream from the DW reach. The RD reach was 960 m long and located immediately downstream of the DW reach. To increase spatial resolution within the DW reach, it was divided into six 820 m subreaches from upstream (DW1) to downstream (DW6). Squirrel Springs enters Snake Creek in DW4. This and other groundwater discharge points can provide significant hydrologic inputs that may influence subreaches DW4 through DW6. Examples of images of some of the channel and valley-bottom variability is illustrated in (Figure 12.2).

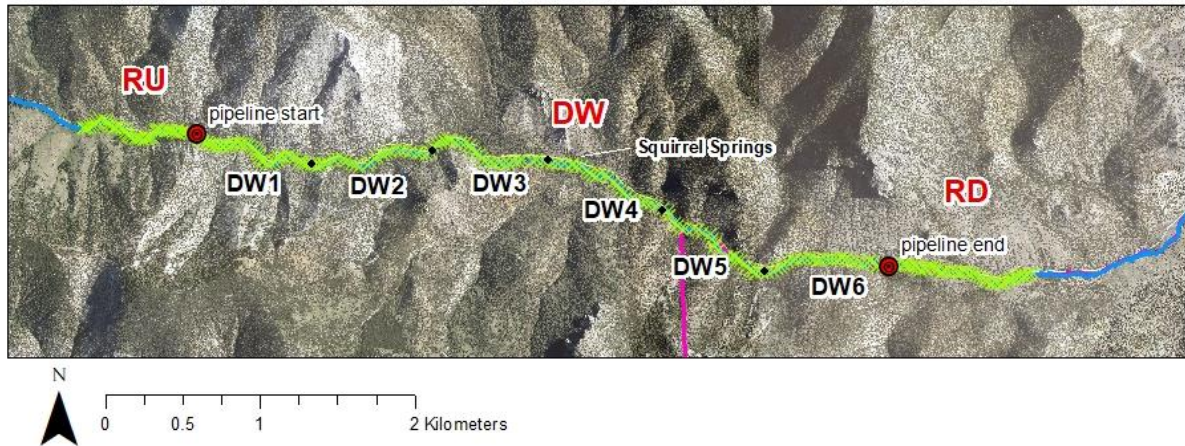


Figure 12.1. The Snake Creek study reaches (RU, DW, RD) are separated by the pipeline inlet and outlet. The DW reach subreaches (DW1-DW6) are separated by black circles. Flow is from left to right. Channel cross-sections (green x's) are shown over Snake Creek (blue) and the Great Basin National Park boundary (pink).

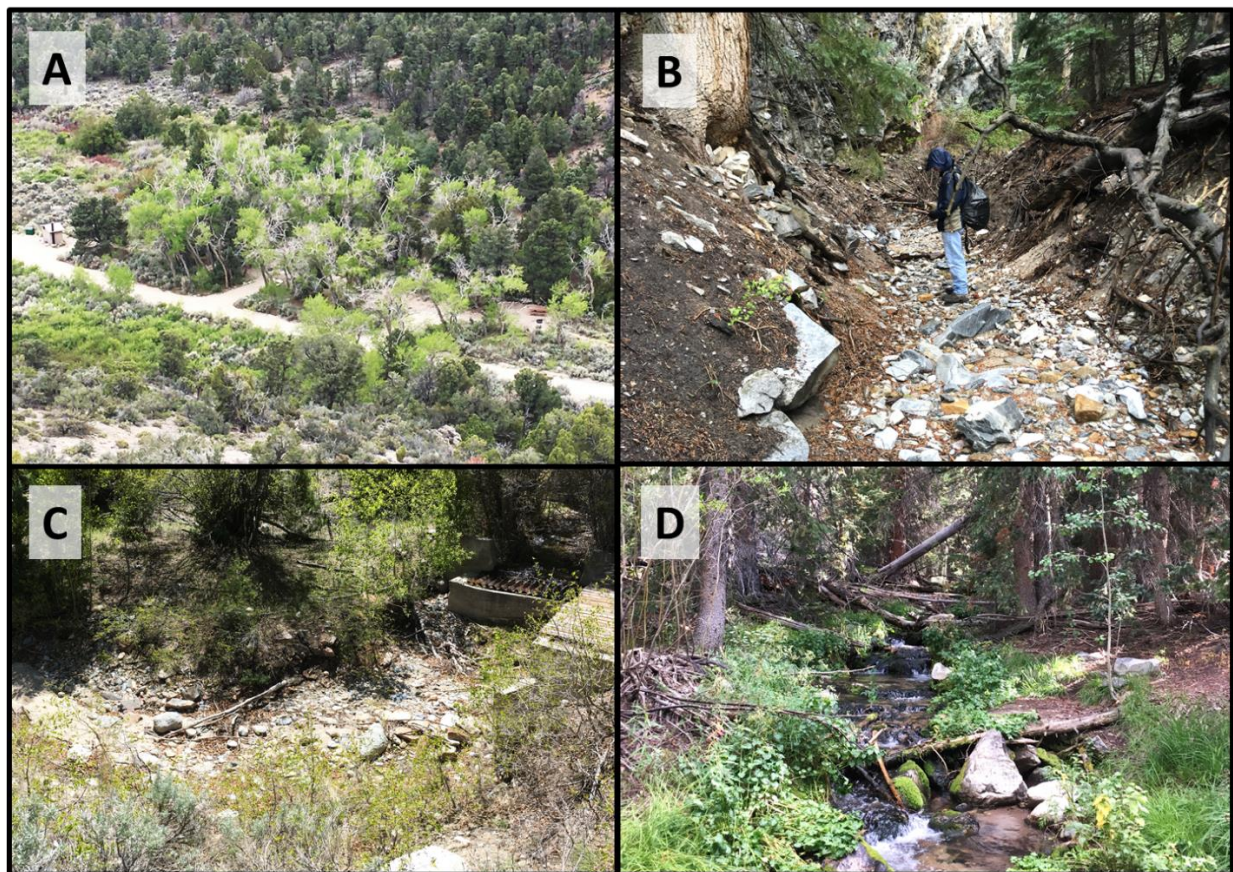


Figure 12.2. Example images of Snake Creek and its valley bottom. (A-C) are from the Dewatered reach, and (D) is from Reference Upstream. The channel is unconfined (A,C,D) or confined (B). See Figure 2.1 for more images of the DW reach channel. (A) is from the Squirrel Springs area depicted in Figure 11.6 and Figure 11.7.

12.2.2 Channel cross-sections

Channel cross-sections were measured approximately every 30 m in DW ($n = 180$; mean $n = 30$ for subreaches) and every 20 m in RU and RD ($n = 44$ and 50). The smaller spacing in reference reaches concentrated their measurements close to DW to facilitate comparison. The active channel was identified at each cross-section by locating a break in slope that coincided with a transition to perennial vegetation (Osterkamp & Hedman, 1982). Partially-confined channels like Snake Creek do not always clearly manifest the active channel, and upstream and downstream morphology were also considered to help identify the active channel at each cross-section. Channel width was measured using either a tape measure or a TruPulse 360 laser range finder (Laser Technology, Inc.). Channel depth was measured from thalweg to active channel shelf using a tape measure. Channel area and width-to-depth ratio were calculated for each cross-section assuming a rectangular channel. Grain size was recorded for each cross-section by either measuring or estimating the secondary axis length of the visually-determined median-sized clast on a cross-section (i.e., D_{50}). This methodology is less rigorous than the common Wolman pebble count, but it enabled a larger sample collection ($n=274$).

We tested for differences in channel width, depth, cross-sectional area, width-to-depth ratio, and grain size by reach. Data were non-normally distributed, so the Kruskal-Wallis rank sum test with post-hoc Dunn tests was used. These tests are a non-parametric alternative to an ANOVA corrected for multiple comparisons. Tests were run in the stats and FSA packages in R, and significance is reported for all $p < 0.05$.

12.2.3 Valley setting

We tested channel width, depth, cross-sectional area, and width-to-depth ratio against the potentially driving variables of valley slope and valley confinement. We used 2008 lidar to create a 1-m digital elevation model (DEM) and determined valley slope by dividing reach length by elevation difference. We used the FluvialCorridor toolbox in ArcGIS to find valley confinement (Roux et al., 2015). This included using the Stream Network tool to generate a Snake Creek flow pathway, the Valley Bottom tool to delineate the valley bottom, the Centerline tool to identify valley centerline, and the Width tool to identify valley bottom width at 20-m intervals. The valley bottom width was defined to include land up to 3 m above the channel thalweg. We produced shapefiles for all layers, including a color-scaled polygon to illustrate valley width (Figure 12.3).

We established relationships between channel dimensions and valley width for cross-sections in the DW reach. Testing within a reach allowed relationships to be identified without the influence of flow diversion differences. We excluded 17 cross-sections around Squirrel Springs because of spring discharge and a history of intense mining disturbance. We also excluded the 7 most-downstream cross-sections because of adjacent spring inflows and because they were shrub-dominated. The remaining 156 cross-sections provided a large sample size to assess channel dimensions as a function of valley width using linear models. Next, we used the linear models to test if differences in channel dimensions across the three reaches were explained by valley width. This involved: 1) identifying if valley width differed using Kruskal-Wallis with post-hoc Dunn tests, 2) using linear regression models to predict the difference in their channel dimensions based on valley widths, and 3) assessing if the models accounted for the measured differences.

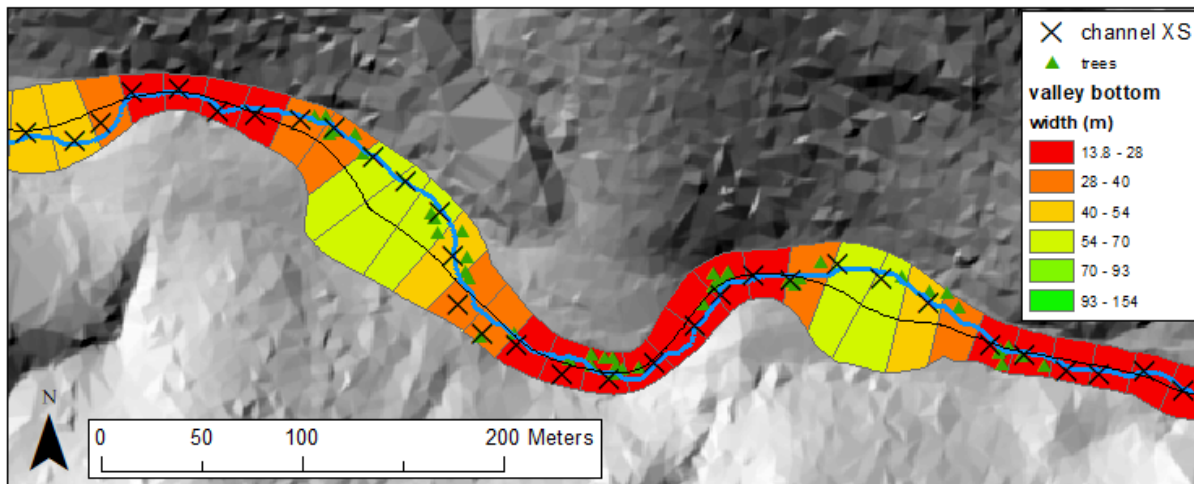


Figure 12.3. An example portion of Snake Creek's DW reach showing 20-meter valley bottom increments. Polygons are color-scaled to depict valley width, with red being more confined and green being wider. Valley centerline is the thin black line, and Snake Creek is the blue line. Measured channel cross-sections (black x's) and surveyed trees (green triangles) are shown.

12.3 Results

12.3.1 Channel cross-sections

Snake Creek channel dimensions varied by reach (Figure 12.4 and Figure 12.5). Kruskal-Wallis with Dunn tests revealed that the DW channel depth of 0.93 ± 0.42 (Std. dev.) m was greater than the equivalently deep reference reaches (p 's < 0.001 between DW and references, $p = 0.62$ between references). Similarly, the DW cross-sectional mean area of 3.59 ± 2.41 m² was larger than that of the reference reaches (p 's < 0.001). The DW channel width of 3.65 ± 1.09 m was greater than RD but statistically similar to RU ($p < 0.001$ and $p = 0.11$). The DW channel width-to-depth ratio was significantly less than RU but similar to RD ($p < 0.001$ and $p = 0.11$). The DW reach had intermediate grain size at 7.6 ± 5.9 cm and did not significantly differ from either reference reach ($p = 0.06$ and 0.10), although RD did have significantly larger bed material than RU ($p = 0.01$).

Dividing DW into subreaches revealed downstream changes in channel dimensions occurring most prominently at the RU-DW1 transition and again between DW3 and DW4 (Figure 12.6). The RU-DW1 transition was located at the start of the pipeline where flow is removed from the channel. Channel width, depth, and area increased here, while the width-to-depth ratio decreased. The DW3-DW4 transition is near Squirrel Springs where the valley widens and there is more frequent Snake Creek flow because of local groundwater discharge. Channel width, depth, and area decreased here, while the width-to-depth ratio increased. These changes essentially reversed the changes at the RU-DW1 transition. Bed grain size generally increased moving downstream, but scatter was large throughout Snake Creek.

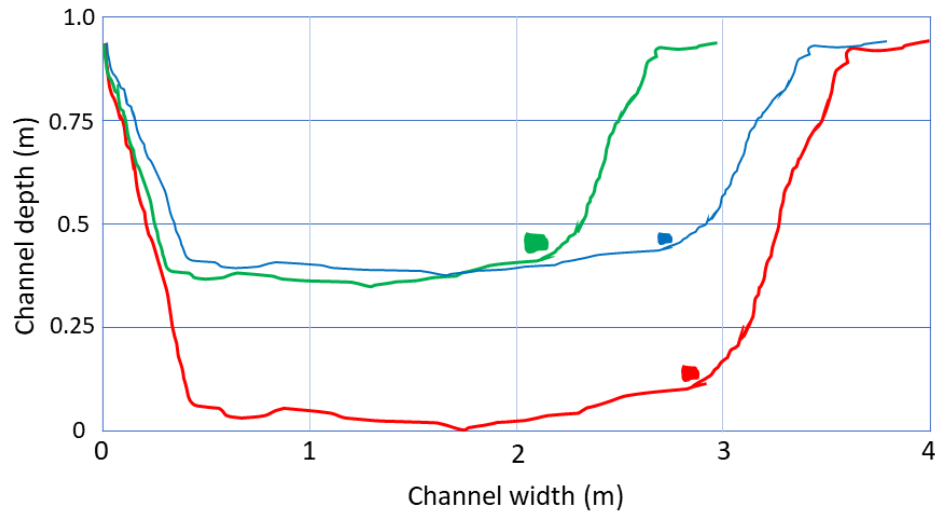


Figure 12.4. Average width, depth, and bed material grain size for the RD (green, smallest cross-section), RU (blue, middle), and DW (red, largest) reaches. Channel dimensions are scaled to 2x vertical exaggeration, and the channel shapes and grain sizes (also scaled to reach average) are idealized for simplicity.

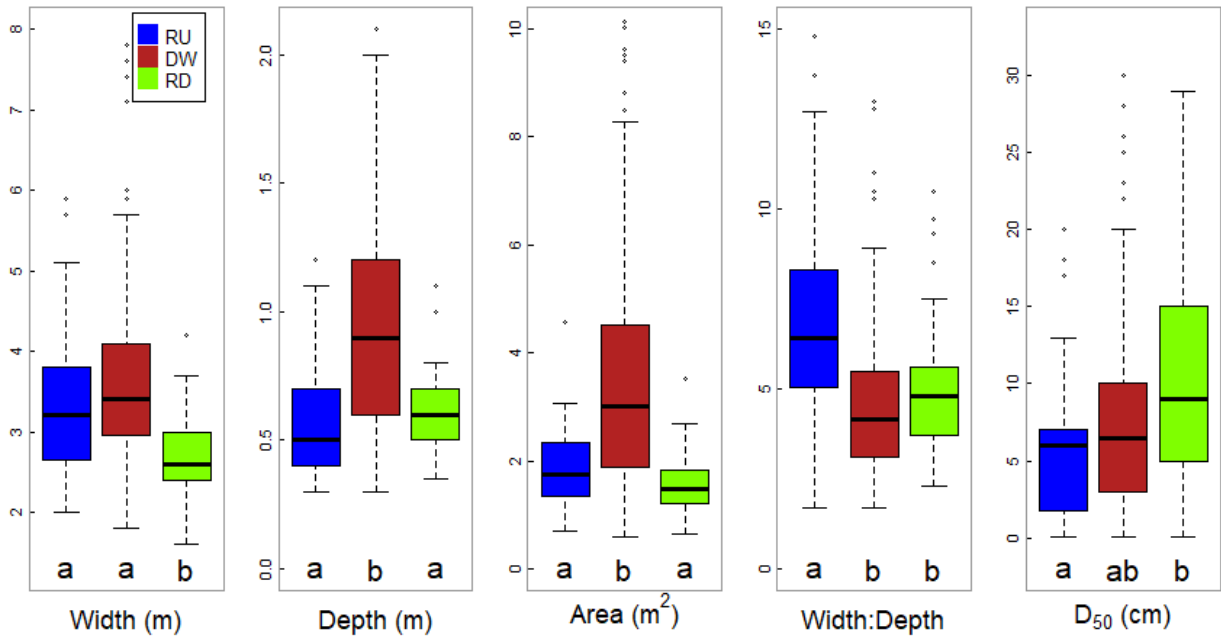


Figure 12.5. Boxplots showing differences in channel dimensions and grain size for the dewatered (DW), reference upstream (RU) and reference downstream (RD) reaches. Significance is displayed by letters, where reaches not sharing a letter differ. For example, “a” differs from “b” but “ab” differs from neither “a” nor “b”.

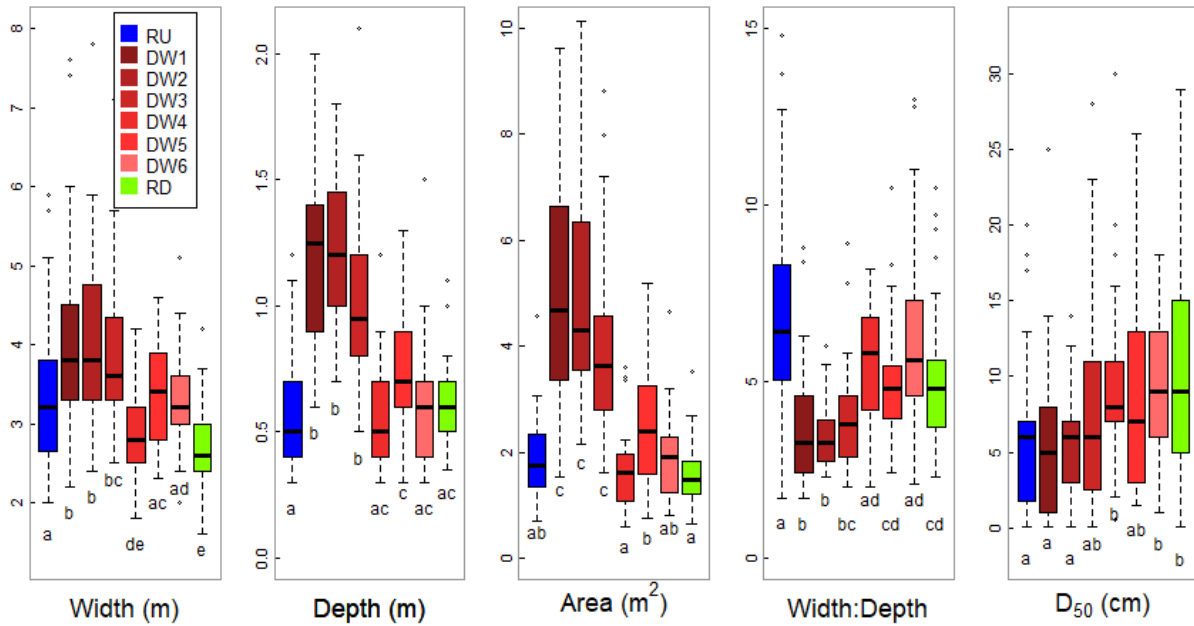


Figure 12.6. Boxplots of channel dimensions and grain size with the study reaches where the DW reach is divided into six subreaches. Squirrel Spring enters Snake Creek in DW4. Significance is displayed by letters where reaches not sharing a letter differ.

12.3.2 Valley setting

Valley slope was similar across reaches: 5.7% in RU, 5.2% in DW, and 5.3% in RD. After dividing DW into six subreaches, the eight stream segments showed the *opposite* relationships of those expected if valley slope drove channel dimensions (Figure 12.7). Higher slopes were associated with lower channel width, depth, and area but higher width-to-depth ratio. For example, DW1 and DW2 had the lowest slopes but the deepest channels, which is opposite of the relationship described by the stream power equation that positively relates erosive capacity and slope. Thus, the data strongly indicated that valley slope did not drive differences in Snake Creek channel dimensions.

The DW reach valley confinement investigation showed valley width to explain a small portion of differences in channel dimensions. The linear models identified all channel dimension variables to be very weakly yet significantly correlated to valley width (p 's < 0.05; $0.03 \leq R^2 \leq 0.10$; Table 12.1; Figure 12.7). Channel width, depth, and area all decreased with increasing valley width, while the width-to-depth ratio increased. These trends supported the hypothesis that increased confinement led to an oversized and incised channel. However, the follow-up analyses described below identified that the relationships did not explain the magnitude of differences.

Valley confinement relationships developed in DW predicted relatively small differences across reaches. DW had the narrowest valley at 39.0 m, while RU was 45.1 m and RD was 57.6 m (Kruskal-Wallis chi-squared = 38.3, $p < 0.001$; Dunn Tests: $2.34 \leq z \leq 5.97$, p 's < 0.05). These valley widths explained a portion of the variation for all four channel dimension variables at RU and three of them at RD (Table 12.2). However, an average of only 21% of the variation was explained. For example, the valley width difference between DW and RD suggested the DW channel should

average 0.21 m wider than the RD channel; however, it averaged 0.98 m wider, so valley confinement explained 21.4% of the difference. Overall, 79% of the variation in channel dimensions was unexplained by valley width. Considering both valley width and the inability of slope to predict channel dimensions, the valley setting explained only a small portion of the measured differences in channel dimensions across study reaches. Therefore, the differences must have been caused by other factors, such as differences in flow.

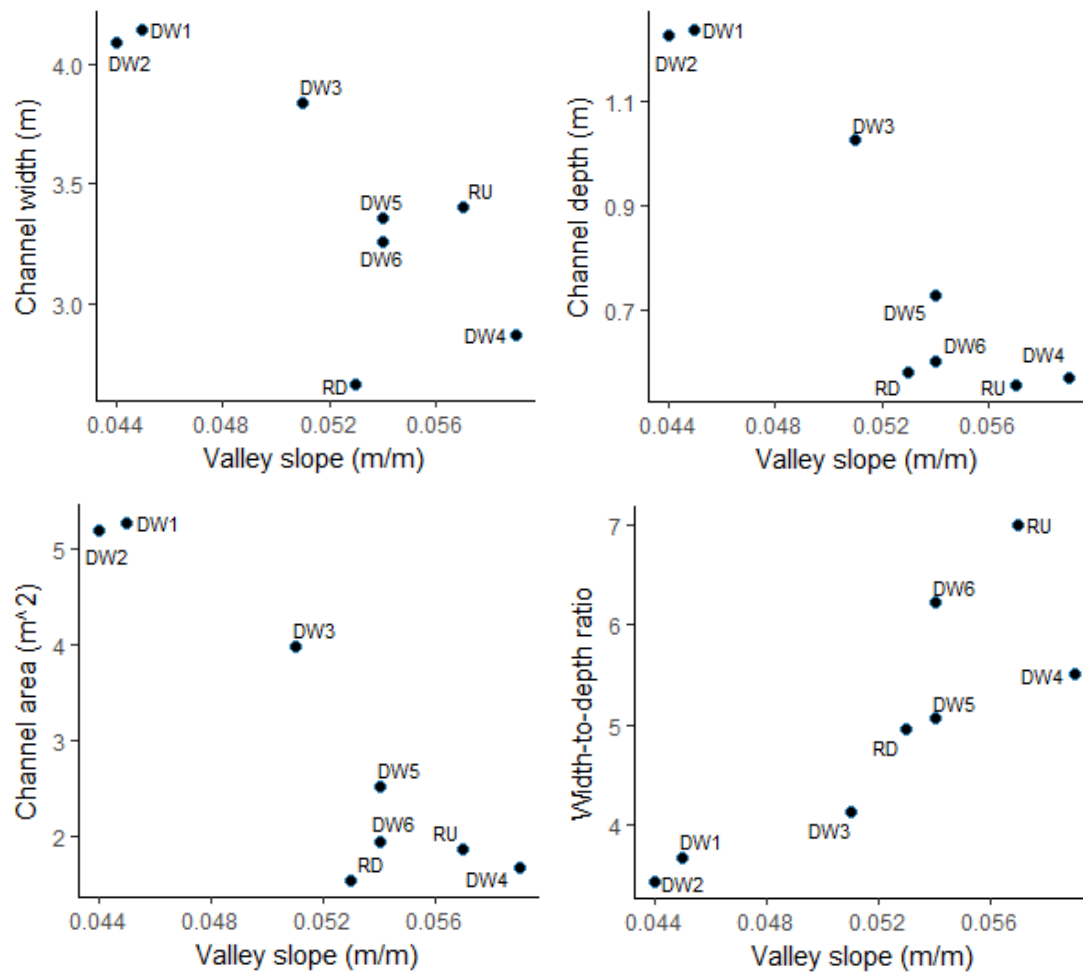


Figure 12.7. Relationships between Snake Creek channel dimensions and valley slope for reference reaches and dewatered subreaches. Note that the relationships for width, depth, and area were *opposite* of those predicted if valley slope dictated channel dimensions. Channel width-to-depth ratio theoretically could vary in either direction with slope depending on hydrogeomorphic setting.

Table 12.1. Linear models used to describe channel dimensions as a function of valley width. Models were calculated from the 156 cross-sections in the DW reach.

Measurement	Slope (m/m)	Intercept (m)	p-value	R ²
Width	-0.011	4.16	0.04	0.03
Depth	-0.008	1.27	< 0.001	0.10
Area	-0.044	5.41	< 0.001	0.08
Width:Depth	0.039	3.09	< 0.001	0.09

Table 12.2. Proportion of the difference in channel dimensions between DW and reference reaches left *unexplained* by valley confinement. Values are means plus/minus 95% confidence intervals. Note that the RD width-to-depth ratio differed from that at DW by 3% in the opposite direction of that expected with valley confinement.

Measurement	RU	RD
Width	0.71 ± 0.27	0.79 ± 0.20
Depth	0.87 ± 0.07	0.57 ± 0.21
Area	0.85 ± 0.08	0.60 ± 0.22
Width:Depth	0.90 ± 0.05	1.03 ± 1.01

12.4 Discussion

The Snake Creek streambed in the DW reach was deeper and had a larger area compared to the two reference reaches. The larger channel is typical of one with the greatest discharge, but stream dewatering leaves it with the lowest discharge after fully removing baseflows and partially removing peak flows. Valley slope and confinement could partially explain conditions, but not to the extent documented. It appears that an external cause unrelated to natural conditions has resulted in an oversized channel in the DW reach.

River flows supports riparian vegetation that promotes bank stability and limits erosion. One way that vegetation protects banks is by providing roughness that deflects energy in high flows (Manners et al., 2015). This energy dissipation decreases the energy available for bed and bank erosion. Additionally, riparian roots provide a physical barrier by deflecting water away from erodible soils and providing structural stability. Dewatering a stream can damage or kill riparian vegetation which decreases bank stability (Amlin & Rood, 2003). The degraded state of riparian vegetation in the DW reach could have increased the erodibility of the banks and led to the modern differences in channel size across reaches (Diehl et al., 2018).

The sediment on the channel bed is a function of lithology, watershed runoff processes, and the ability of past flows to transport the material supplied. Moving water contains energy that transports sediment, and more energy can move increasingly larger clasts. Transport competence also increases

as width-to-depth ratio decreases. This is consistent with the observed pattern of the smaller and wider RU channel having the smallest clasts. In step-pool channels with a relatively consistent slope such as Snake Creek, sediment coarsening may be expected further downstream as watershed area, discharge, stream power increase (Lane, 1954). This expectation was generally supported (Figure 12.5, last panel), although high clast size variability resulted in few significance trends among the subreaches (Figure 12.6, last panel). The complex lithology, high infiltration rates, and relatively recent stream dewatering have not resulted in a clear effect on sediment size distribution.

Oversized channels like Snake Creek in much of the DW reach resemble ephemeral streams and washes that flow too infrequently to sustain riparian vegetation (Gonzalez, 2001; Kamp et al., 2013). In these streams, channel width, depth, and area may not be proportional to average flow conditions. Instead, their dimensions result from infrequent large-scale events such as peak spring runoff and monsoonal rainstorms that generate high runoff. The large size of Snake Creek in the DW reach is suggestive of such conditions, which are related to bank instability and slope failures previously documented (Van Hoesen, 2002). The channel slope was similar among the upstream (4.8%) and downstream (5.5%) reference reaches compared to all DW subreaches (mean 5.3%, range of 4.5–6.0%). While valley-setting hypotheses were partially supported by valley confinement, all measurements suggest that another factor has led to the dewatered portion of the Snake Creek streambed being oversized and unstable.

13. Instream wood

13.1 Introduction

Wood can be incorporated into river channels as trees die or banks erode. Instream wood often benefits rivers and streams by increasing channel complexity and habitat diversity, but unnaturally high wood loads can cause bank destabilization and disrupt the discharge balance between water and sediment (Wohl, 2001). We compared wood loads in the Snake Creek DW reach to reference reaches immediately upstream and downstream (RU and RD). We hypothesized that the DW reach would have more instream wood because of its high tree mortality and lower bank stability. This instream wood analysis complemented the cottonwood mortality surveys (Section 3.4) by providing an independent approach to quantifying riparian forest dieback and describing a role of wood after tree death. The analysis complemented channel morphology measurements by investigating dead trees as potential causes and effects of the oversized channel found in the DW reach (Section 12.3.1).

13.2 Methods

We measured instream wood loads in Snake Creek in August 2017 (Figure 13.1). The focal reach was again the DW reach where Snake Creek flow is diverted into the pipeline. The upstream and downstream reference reaches matched those from the channel cross-section analysis. The RU reach extended 870 m up from the pipeline inlet and included all of the RU cottonwood reach. The RD reach extended 960 m down from the pipeline outlet and included all of the RD cottonwood reach.

We measured all instream wood jams greater than 0.2 m x 0.2 m x 1 m long. These wood jams were one or more pieces of wood piled together on the channel bed. Jams were distinct from each other if they did not touch. We measured or estimated jam length, width, and height to create a rectangular volume of jam extent. Height was measured with a tape measure to decimeter precision, while width and length extended up to 25 meters and were usually measured to decimeter precision with a laser range finder. For every jam we estimated wood density as the percent of the jam's rectangular volume that was wood and not air, sediment, or detritus. This estimate was not robustly calibrated due to its complexity, but assessment was consistent across reaches. Multiplying these four measurements produced a volume of wood for each jam. We also recorded if a jam spanned the channel width and if there were clear signs of erosion or deposition associated with it. Wood in jams was not identified to species, so jams represent dieback of all trees including cottonwood, aspen, birch, juniper, white fir, and pinyon pine. However, most wood was from cottonwoods.

We calculated the mean and standard deviation for jam width, height, length, density, and volume for the study reaches. Skewed distributions led us to calculate reach differences using the non-parametric Kruskal-Wallis test with follow-up Dunn tests to identify significant differences between reaches. We used reach length to calculate jam frequency and wood load per kilometer.



Figure 13.1. Examples of instream wood jams in the DW (panels A and B), RU (panel C), and RD (panel D) reaches.

13.3 Results

Instream wood loads varied within and across Snake Creek reaches (Figure 13.2). Total instream wood was more abundant and jams were larger in DW compared to RU and RD. There was $61.2 \text{ m}^3/\text{km}$ of wood in the DW reach compared to $23.0 \text{ m}^3/\text{km}$ in RU and $10.6 \text{ m}^3/\text{km}$ in RD. Mean jam width, height, length, volume, and density were all highest at DW (Table 13.1). Jam width and height had the same rankings as channel width and height for the three study reaches. This suggested some causal relationship exists between channel size and jam size, although mechanisms were not tested. Individual jams contained an average of 1.78 ± 3.34 (standard deviation) m^3 of wood in the DW reach, which was 4–5 times greater than the two reference reaches (Figure 13.3; Dunn's $Z = 3.3\text{--}3.6$, p 's ≤ 0.001). Other significant relationships included jams being wider in DW than RD, taller in DW than RU, and longer in DW than both RU and RD.

Although wood load and jam size were greatest in DW, many aspects of the jams were similar across the study reaches. Jam frequency was somewhat similar at 31 jams/km in RD, 35 jams/km in DW, and 48 jams/km in RU. In the three reaches, 93–97% of jams spanned the channel width and 53–59% caused sediment deposition. Bed and bank erosion varied somewhat more and ranged from a low of 33% of jams causing erosion in RU to a high of 53% in RD (Table 13.1).

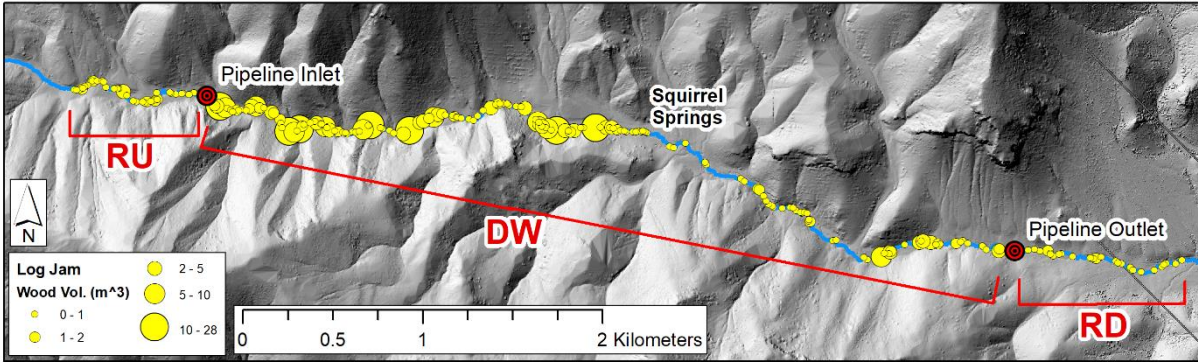


Figure 13.2. Instream wood jams in the three Snake Creek study reaches. Each jam is a yellow circle scaled to the volume of wood it contained. Snake Creek is blue and flow is from left to right.

Table 13.1. Wood jam characteristics for the DW, RU, and RD reaches. Superscript “U” and “D” show significant difference at $p < 0.05$ between DW and RU or RD, respectively.

Measurement	RU	DW	RD
Number of jams	39	169	30
Jam frequency (km^{-1})	47.6	34.5	31.3
Channel spanning (%)	97	93	93
Erosional (%)	33	47	53
Depositional (%)	56	59	53
Width (m)	3.05 ± 0.85	3.36 ± 1.36^D	2.78 ± 0.71
Height (m)	0.57 ± 0.24	0.75 ± 0.34^U	0.60 ± 0.20
Length (m)	2.45 ± 4.29	3.27 ± 3.73^{UD}	1.46 ± 1.47
Wood density (%wood/jam vol.)	18.0 ± 13.0	19.7 ± 15.9	16.4 ± 11.7
Wood per jam (m^3)	0.48 ± 0.58	1.78 ± 3.34^{UD}	0.34 ± 0.34
Wood per river km (m^3)	23.0	61.2	10.6

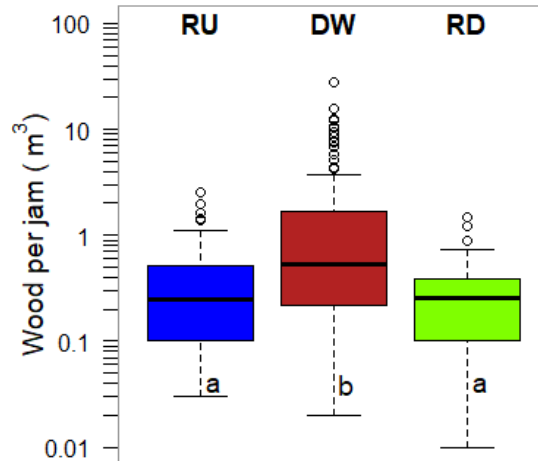


Figure 13.3. Volume of wood per jam in reference upstream (RU), dewatered (DW), and reference downstream (RD) reaches. Lower-case letters show that jam size was significantly larger in DW compared to both reference reaches. Note the log y-axis.

13.4 Discussion

Instream wood distribution revealed one important way that Snake Creek's DW reach differed from reference reaches. Although the frequency of DW wood jams was intermediate to the two reference reaches, their larger size caused the wood load to be highest in the DW reach by 3 to 6 times. The larger instream wood volume caused larger, not more, jams. This could have been because the larger channel cross-section in the reach (Section 12.3.1), along with its less streamside vegetation, facilitated downstream wood transport. Wood loads were greatest in the upper portions of the DW reach, where channel dimensions were also greatest. Instream wood measurements supported both the hypothesis that dying trees have deflected flow and led to a larger channel, and the hypothesis that an enlarged channel led to increased instream wood recruitment.

We did not analyze the downstream variability of wood *within* the DW reach, but it clearly varied (Figure 13.2). Wood jams in the upper parts of DW were largest, and there was little instream wood below Squirrel Springs and in the Reference Squirrel subreach of DW, which supported apparently healthy riparian vegetation with a high proportion of living cottonwood canopy. The generally higher valley confinement in the upper portion of DW allowed more falling trees to reach the channel. A second factor is that cottonwood death occurred earlier in the upper half of DW (Figure 3.9). Earlier tree mortality gave more time for dead trees to fall, again meaning that instream wood supply would increase higher in the DW reach. The heterogeneity identified in our instream wood analysis provides more evidence that complex biological and physical interactions underpin the Snake Creek riparian ecosystem.

Although the shorter length and resulting fewer jams in the two reference reaches produced jam dimensions that were statistically similar to each other, the RU and RD jams differed in interpretable ways. RU jams were more frequent, longer, and had more wood. This led to an instream wood load 1.4x greater per kilometer in RU than RD. This difference is consistent with the RU cottonwood

forest being older (Figure 3.4) and presumably more likely to die, fall, and produce instream wood. RU trees were anecdotally more densely spaced and taller, which also increased their potential to create instream wood. Higher elevation channels like the RU reach can have a greater proportion of snowmelt runoff and fewer convective thunderstorms (Clow, 2010; Kampf & Lefsky, 2016). The resulting more consistent flow regime should support riparian vegetation and stabilizes channel banks. This scenario could explain why even though jams were larger at RU, they were less frequently erosive if its dense riparian vegetation stabilized banks. Even with the differences identified, RU and RD jams varied less than their common difference from DW. Ultimately, the anomalies from the DW reach, which was located between the reference reaches, suggest that DW conditions differed from those of stable portions of Snake Creek and the historical norm.

14. Summary and Synthesis

This study used 11 research projects to investigate the physical and biological conditions of the Snake Creek riparian ecosystem (report Sections 3 to 13). The key results from each project are summarized below before being comprehensively synthesized.

14.1 Summary of each research investigation

Section 3: Cottonwood forest condition: Demography, canopy vigor, and tree mortality. We characterized cottonwood growth and death to describe the age structure, dieback, and stability of the riparian forest across the study reaches. All reaches were dominated by trees of older age classes. There has been little regeneration in recent decades, and successful recruitment events are needed in the future decades to replace the old and dying trees. Accelerated tree death, such as we found in the DW reach, increases forest stand vulnerability. Dead cottonwoods were 2.2 to 2.6 times more common in the dewatered (DW) reach than in reference reaches, and the percent canopy in live trees was 43% in the DW reach and 63% to 76% in reference reaches. Leaf area measurements made with a ceptometer supported the visually assigned canopy classification, as did the NDVI aerial photo analysis. The low percent live canopy was not caused by old tree age because the very-old and moderately-old cohorts in the DW reach both had similar canopy vigors. Trees in the DW reach had the fastest decline in tree ring growth before death, suggesting that growing conditions there changed most abruptly. We found three main periods of DW cottonwood mortality: 1968–1974, 1986–1992, and the largest in 2005–2014. However, the within-tree data showed that branch mortality was highest in the 1960s-1970s, and the median branch death date was 20 years before the whole tree died. This suggests that trees experienced a prolonged decline preceding death.

Section 4: Tree ring chronologies. Tree ring chronologies developed from annual growth rings revealed spatial and temporal variability in growing conditions across the study area. Ring widths in DW trees declined in the 1960s and 1970s, rebounded in the wet 1980s, and declined since then having unprecedented declines since the 2000s. Relative to reference reach chronologies, low-growth periods began in DW trees in the 1960s following dewatering and again in the 2000s. DW reach cottonwood decline was not caused by old tree age. Subreach analysis revealed high variability in growth patterns within the DW reach, illustrating a complex growth environment over the 5-km reach. Since the year 2000, tree ring growth was perfectly ordered with percent living canopy, as trees with more live canopy had wider rings. Trees that died had the narrowest rings of any canopy class in the years preceding their death, although their annual growth patterns had the same relative highs and lows as living trees. It is likely that high mortality in the DW reach will continue if dewatered conditions persist.

Section 5: Stable isotopes in tree rings. Stable carbon isotopes in tree rings can be used to identify periods of water stress at the time of carbon assimilation because stomatal closure causes isotopic enrichment of ^{13}C over ^{12}C . Snake Creek cottonwood $\delta^{13}\text{C}$ values (i.e., $^{12}\text{C}/^{13}\text{C}$ ratio) abruptly increased in the early 1960s in DW reach trees but not reference reach trees, indicating drought conditions were experienced by the DW reach trees. The $\delta^{13}\text{C}$ values briefly returned to baseline levels during the wet mid-1980s, but higher values returned and persisted through the end of our

study period in 2016. $\delta^{13}\text{C}$ was most strongly correlated with streamflow and precipitation for DW reach trees, suggesting that drought was greatest in the DW reach. A cluster analysis perfectly separated the study trees into two groups, one with DW reach trees and another with trees from the upstream and downstream reference reaches, even though the two reference reaches were spatially separated by the DW reach. $\delta^{13}\text{C}$ in DW reach trees was negatively correlated with tree ring width, which is known to occur in other dry settings. In contrast, a weak positive relationship was found between the two variables in reference reach trees, indicating that water stress had less influence on their productivity.

Section 6: Hydroclimatic drivers of tree growth. We linked tree ring width to streamflow, precipitation, and air temperature to identify the climate variables most limiting to tree growth and to see if these variables were stable through time. Late-winter and spring growing conditions had the strongest correlations to tree ring width for all three climate variables, suggesting that these seasons are most pivotal to forest productivity. Tree ring width was positively correlated with flow and precipitation in all five study reaches but negatively correlated with temperature. This indicated that low water access and high temperature, which occur during drought conditions, inhibited cottonwood growth everywhere. The correlations with flow and precipitation were strongest in the DW reach, however, indicating that water most limited tree growth in the DW reach adjacent to the pipeline. The pre- vs. post-pipeline analysis revealed increasing correlations between tree rings and water availability at DW but weakening correlations at reference reaches, indicating that during the pipeline period water was most limiting to growth in the DW reach.

Section 7: Ecophysiology. We characterized cottonwood water transport capacity to investigate if chronic water stress led to physiologically interpretable differences between trees in the DW and reference downstream (RD) reaches. We found small differences in cottonwood leaf water potential between reaches. Trees in the DW reach went to slightly lower water potentials during the midday stressful period. Experiments to measure loss of leaf hydraulic conductance revealed that DW trees photosynthesized less than RD trees under the relatively dry conditions encountered in a typical day. This may have been because the less reliable water in the DW reach caused them to be more conservative under more risky conditions. However, the small differences between the two reaches indicated that xylem water potential adjustments were less substantial than other adjustments, such as branch sacrifice and stomatal closure leading to isotope enrichment.

Section 8: Aerial photos. Analysis of historical aerial photographs was used to detect the expansion or contraction of the Snake Creek riparian forest from 1946 to 2017. We found that riparian forest area declined from 10.2 to 5.6 hectares (25.2 to 13.9 acres) in the DW reach while it expanded slightly from 8.6 to 9.1 hectares (21.1 to 22.6 acres) in the reference reaches. The decline began in the first photo of the post-pipeline period, then vegetated area increased during the wet 1980s. However, after 1984 riparian area declined in the DW reach but not the reference reaches. Individual trees with successively higher percentages of live canopy also had successively higher NDVI values. Reference reach trees had higher NDVI values than DW reach trees, indicating that reference areas had healthier vegetation from 1981 to 2017. The lower resolution, intermittent frequency, and role of other landscape factors made it impossible to identify exactly when the change began in the DW

reach, but our analysis indicated that trees in the two reaches had different trajectories that increasingly deviated through 2017.

Section 9: Stand-level cottonwood production. Two complementary approaches were used to characterize whole-forest growth for assessment of whole-forest production. We combined annual tree ring growth with either aerial photos or tree counts to quantify cottonwood basal area production standardized to stream length. We identified a decline in the DW reach riparian forest production relative to its pre-pipeline values and to reference reach values. The aerial photo approach indicated that DW riparian forest production was 67% of its baseline during the pipeline period, and cottonwood basal production declined by $31 \text{ cm}^2/\text{km}/\text{yr}$ during the pipeline period. In contrast, the reference reaches maintained 97% of pre-pipeline growth and declined by only $0.4 \text{ cm}^2/\text{km}/\text{yr}$. The tree-count approach showed RD production to be stable through time, while DW and RU had declines for most of the study period that became most extreme at DW in the final decade of study. Overall, the cottonwood forest declined more in the DW reach than in reference reaches.

Section 10: Groundwater monitoring. Depth to water in eight groundwater monitoring wells and river stage at five stream gauges revealed strong relationships between surface water and groundwater in the Snake Creek riparian corridor. Between the time of well installation in July 2018 and snowmelt runoff that began in April 2019, the DW reach had no surface water flow and groundwater was present within 5 m of the ground surface in only one of the four areas measured. This indicated that groundwater was likely too deep to be available to most cottonwoods. After the unusually high 2019 snowpack melted, water flowed through the entire DW reach for at least 78 days in the growing season. Squirrel Springs was a clear inflection point in surface and subsurface hydrologic processes. The upper half of the DW reach had a strong losing gradient of surface water to the groundwater. Below Squirrel Springs surface water continued to recharge groundwater, but a relatively neutral surface water-groundwater gradient indicated that little water was lost to deep recharge. Water level data from below the pipeline outlet in the RD reach showed perennial surface water connected to groundwater that ranged from 0.6 to 2.0 m below ground, within the cottonwood root zone. Beyond hydrologic data, well drilling revealed highly variable lithology within the Snake Creek valley bottom.

Section 11: Near-surface geophysics. Subsurface conditions near Snake Creek were interpreted using electrical resistivity imaging to identify geologic contacts, zones of saturation, and hydrologic relationships between the channel and alluvial groundwater. Electrical resistivity profiles indicated that the subsurface was substantially drier in the DW reach than in reference reaches. The water table had a negative slope from Snake Creek to the adjacent riparian zone in all areas studied, indicating that surface water flowed from the stream into the subsurface where it was accessible to riparian vegetation. Immediately downstream of the pipeline inlet, saturated conditions dropped to approximately 16 m below ground. The water table inferred from all DW reach profiles was deeper than could be accessed by cottonwood roots. These depths would likely kill cottonwoods if water was not more accessible at other times of year or in high runoff years such as 2019. Cottonwood distribution was narrowly confined to the near-channel zone in the DW reach where the water table was at least 5 m below the channel bottom. The riparian forest widened in areas where groundwater

discharged into the valley bottom, indicating that alluvial groundwater access is essential to maintain or restore the riparian forest.

Section 12: Channel and valley morphology. Channel dimensions are influenced by a variety of factors including valley width and slope, flow regime, riparian vegetation, and wood jams. Different channel dimensions along Snake Creek revealed that some of these factors differed along the river and modified geomorphic conditions. We found the Snake Creek channel to be wider, deeper, and having a larger cross-sectional area in the DW reach compared to the RU and RD reference reaches. Separating the DW reach into subreaches showed that the unusually large channel dimensions were concentrated in the upper half of the reach, above the influence of Squirrel Springs and other groundwater discharge points. Although valley width partially explained channel dimensions, the differences exceeded those explainable by any topographic factor such as valley width and slope. Instead, differences in flow regimes and riparian vegetation vigor more likely caused the differences.

Section 13: Instream wood. Dying riparian trees that fall into the channel can increase instream wood loads and destabilize channel banks, resulting in larger channels that can become disconnected from riparian plants. The instream wood load was 3 to 6 times greater in Snake Creek's DW reach than in the RU or RD reaches. Although the frequency of wood jams was similar across reaches, their greater width, height, and length caused wood volume per jam to be 4 to 5 times greater in the DW than reference reaches. Most instream wood was from dead cottonwoods and located in the upper half of the DW reach, where trees died earlier and channel dimensions were largest. This geomorphic view of the dead riparian trees expands the perspective on how changes to water access have cascaded through Snake Creek's riparian ecosystem.

14.2 Synthesis

Stream dewatering has altered the physical and biological conditions in the Snake Creek riparian corridor. In the DW reach the stream channel is dry most of the year, and alluvial groundwater is generally deeper than is accessible to riparian plants. The proportion of dead cottonwoods in the DW reach was 2.4 times higher than in reference reaches, and its surviving trees had only 60% as much live canopy as reference reach trees. Elsewhere in western North America, water diversions and dams have led to riparian forest collapse (Grismer, 2018; Rood & Heinze-Milne, 1989; Rood & Mahoney, 1990; Shafroth et al., 2002). Forest collapse is ongoing in the Snake Creek DW reach, where base flows and moderate runoff are removed and flow passes the diversion only during floods. These ephemeral flows and local precipitation have allowed some cottonwoods to survive decades of stream dewatering. However, atmospheric drought and ongoing hydrologic stress have largely exceeded the tolerance of surviving trees, leading to an ecosystem decline.

Multiple research approaches were used to reconstruct the causes and timing of environmental changes along Snake Creek, revealing that DW reach cottonwoods have increasingly diverged from reference conditions. Stable carbon isotopes in tree rings indicated an abrupt onset of low-moisture stress at the time of pipeline construction in 1961 that occurred only in the DW reach. Pulses of whole tree and branch death beginning in the 1960s likely represent initial forest adjustments to reduced water availability. Branches had prolonged dieback and preceded death of the whole tree by an average of 20 years. Cottonwoods are known to sacrifice branches to reduce leaf water losses

during drought conditions (Rood et al., 2000; Schook et al., 2016a), but branch sacrifice also reduces photosynthesis and can lead to tree death via starvation (Allen et al., 2015).

The correlation of ring width to flow and precipitation in the post-diversion period was greater in the DW reach than in any reference reach, indicating the stronger limitation of growth by water shortage resulting from flow diversion. Ring width chronologies revealed tree growth declines in the DW reach during the 1960s and the mid-2000s. We suspect that the first decline was an immediate adjustment to flow diversion and the second resulted from increased susceptibility to atmospheric drought after decades of water stress. By 2016, the DW reach had the lowest growth rate of any reach, even though its trees had grown 1.4 to 2.1 times faster than reference trees in the 1901–1961 pre-diversion period. Tree ring growth rates mirrored the proportion of live tree canopy, demonstrating that outward appearance reflected internal conditions related to carbon assimilation. Thus, visual observations of poor tree health by park managers and visitors were reliable indicators of tree health.

We combined aerial photography, ring widths, and a census of live and dead trees to document effects of flow diversion on productivity of the entire cottonwood forest in Snake Creek's DW and reference reaches. Forest productivity in the DW reach has declined steadily since diversion, except for a temporary increase during the wet 1980s. Productivity is now lower in the DW reach than in any reference reach. Aerial photography indicates that the cottonwood canopy has fragmented following tree death and branch dieback, reducing the riparian forest area to 55% of its pre-pipeline average.

The health of Snake Creek's riparian forest is partly determined at subreach scales that vary with hydrogeomorphic setting, as illustrated *within* the DW reach. Most DW reach cottonwoods are adjacent to the channel in the Snake Creek subreach above Squirrel Springs, where monitoring wells revealed a strongly losing water table that rapidly deepened with distance from the channel. In contrast, there was a wider forest downstream of Squirrel Springs where the water table was nearer to the ground surface across much of the valley bottom and some trees had access to spring groundwater discharges. Trees in the upstream subreach died earlier and many had fallen to the ground, contributing to the larger wood jams and larger channel cross-sections documented there. A complex feedback appears to have occurred, with flow diversion reducing water access and killing trees, which then fell into the channel and destabilized banks by deflecting hydraulic forces into the formerly vegetation-stabilized sediments. Relationships such as this can help to explain why dewatering effects were less clear prior to our focused investigation.

The highest surface water runoff in the past 58 years and the highest precipitation in 100 years occurred in the early 1980s. During this wet period Snake Creek's flow would have exceeded the pipeline's intake capacity, allowing the creek to flow through the DW reach and replenish the alluvial groundwater and soil water accessed by riparian plants. Every analysis we conducted indicated that the riparian forest decline temporarily halted during this period. Aerial photos indicated a slight riparian area expansion and increased forest stand production, and few branches or trees died at this time. Carbon isotope trends temporarily reversed, and tree rings were the widest of

the past century. This climatically wet period can be considered a natural experiment partially resembling ecosystem responses if flow diversion ceased and water was returned to the stream.

Recovery of the degraded ecosystem could take many decades, as the cottonwood forest cannot recover without new tree establishment. However, some components and processes are likely to respond rapidly if natural runoff is reinstated. Hydrological, ecological, and physiological changes caused by flow modification have been reversed elsewhere with hydrologic restoration (Amlin & Rood, 2003; Díaz-Redondo et al., 2018; Rood et al., 2003; Stromberg, 2001). Beyond the cottonwood forest, regaining natural flows could help re-establish Bonneville cutthroat trout, aquatic invertebrates, structurally and biologically diverse riparian communities, and food webs reliant on aquatic-terrestrial exchanges (Nakano et al., 1999; Sabo & Power, 2002). Additionally, by increasing understanding of ecosystem processes through scientific study, it will be possible to make more informed water management decisions in this region where water access is crucial.

15. References

- Albert, C., Schröter, B., Haase, D., Brillinger, M., Henze, J., Herrmann, S., et al. (2019). Addressing societal challenges through nature-based solutions: How can landscape planning and governance research contribute? *Landscape and Urban Planning*, 182, 12–21. <https://doi.org/10.1016/j.landurbplan.2018.10.003>
- Albertson, F. W., & Weaver, J. E. (1945). Injury and Death or Recovery of Trees in Prairie Climate. *Ecological Monographs*, 15(4), 393–433. <https://doi.org/10.2307/1948428>
- Allen, C. D., Breshears, D. D., & McDowell, N. G. (2015). On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene. *Ecosphere*, 6(8), 1–55.
- Allen, D. C., Wynn-Thompson, T. M., Kopp, D. A., & Cardinale, B. J. (2018). Riparian plant biodiversity reduces stream channel migration rates in three rivers in Michigan, U.S.A. *Ecohydrology*, 11(4), e1972. <https://doi.org/10.1002/eco.1972>
- Amlin, N. M., & Rood, S. B. (2003). Drought stress and recovery of riparian cottonwoods due to water table alteration along Willow Creek, Alberta. *Trees*, 17(4), 351–358. <https://doi.org/10.1007/s00468-003-0245-3>
- Anchukaitis, K. J., Evans, M. N., Lange, T., Smith, D. R., Leavitt, S. W., & Schrag, D. P. (2008). Consequences of a Rapid Cellulose Extraction Technique for Oxygen Isotope and Radiocarbon Analyses. *Analytical Chemistry*, 80(6), 2035–2041. <https://doi.org/10.1021/ac7020272>
- Andersen, D. C. (2015). Tree Mortality in Mature Riparian Forest: Implications for Fremont Cottonwood Conservation in the American Southwest. *Western North American Naturalist*, 75(2), 157–169. <https://doi.org/10.3398/064.075.0204>
- Anderson, D. B. (1936). Relative humidity or vapor pressure deficit. *Ecology*, 17(2), 277–282.
- Attia, Z., Domec, J.-C., Oren, R., Way, D. A., & Moshelion, M. (2015). Growth and physiological responses of isohydric and anisohydric poplars to drought. *Journal of Experimental Botany*, 66(14), 4373–4381. <https://doi.org/10.1093/jxb/erv195>
- Au, R., & Tardif, J. C. (2012). Drought signals inferred from ring-width and stable carbon isotope chronologies from Thuja occidentalis trees growing at their northwestern distribution limit, central Canada. *Canadian Journal of Forest Research*, 42(3), 517–531. <https://doi.org/10.1139/x2012-012>
- Baker, D. W., Bledsoe, B. P., Albano, C. M., & Poff, N. L. (2011). Downstream effects of diversion dams on sediment and hydraulic conditions of Rocky Mountain streams. *River Research and Applications*, 27(3), 388–401. <https://doi.org/10.1002/rra.1376>

- Beechie, T. J., Liermann, M., Pollock, M. M., Baker, S., & Davies, J. (2006). Channel pattern and river-floodplain dynamics in forested mountain river systems. *Geomorphology*, 78(1–2), 124–141. <https://doi.org/10.1016/j.geomorph.2006.01.030>
- Bigler, C., & Bugmann, H. (2003). Growth-dependent tree mortality models based on tree rings. *Canadian Journal of Forest Research*, 33(2), 210–221. <https://doi.org/10.1139/x02-180>
- Bigler, C., & Bugmann, H. (2004). Predicting the time of tree death using dendrochronological data. *Ecological Applications*, 14(3), 902–914. <https://doi.org/10.1890/03-5011>
- Biondi, F. (1999). Comparing Tree-Ring Chronologies and Repeated Timber Inventories as Forest Monitoring Tools. *Ecological Applications*, 9(1), 216–227. [https://doi.org/10.1890/1051-0761\(1999\)009\[0216:CTRCAR\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1999)009[0216:CTRCAR]2.0.CO;2)
- Biondi, F., & Qeadan, F. (2008). A Theory-Driven Approach to Tree-Ring Standardization: Defining the Biological Trend from Expected Basal Area Increment. *Tree-Ring Research*, 64(2), 81–96. <https://doi.org/10.3959/2008-6.1>
- Blaschke, T. (2010). Object based image analysis for remote sensing. *ISPRS Journal of Photogrammetry and Remote Sensing*, 65(1), 2–16. <https://doi.org/10.1016/j.isprsjprs.2009.06.004>
- Brendel, O., Iannetta, P. P. M., & Stewart, D. (2000). A rapid and simple method to isolate pure alpha-cellulose. *Phytochemical Analysis*, 11(1), 7–10. [https://doi.org/10.1002/\(SICI\)1099-1565\(200001/02\)11:1<7::AID-PCA488>3.0.CO;2-U](https://doi.org/10.1002/(SICI)1099-1565(200001/02)11:1<7::AID-PCA488>3.0.CO;2-U)
- Brienen, R. J. W., Gloor, E., & Zuidema, P. A. (2012). Detecting evidence for CO₂ fertilization from tree ring studies: The potential role of sampling biases. *Global Biogeochemical Cycles*, 26(1), GB1025. <https://doi.org/10.1029/2011GB004143>
- Briffa, K. R., Jones, P. D., Bartholin, T. S., Eckstein, D., Schweingruber, F. H., Karlén, W., et al. (1992). Fennoscandian summers from AD 500: temperature changes on short and long timescales. *Climate Dynamics*, 7(3), 111–119. <https://doi.org/10.1007/BF00211153>
- Briffa, Keith R., & Melvin, T. M. (2011). A closer look at regional curve standardization of tree-ring records: justification of the need, a warning of some pitfalls, and suggested improvements in its application. In *Dendroclimatology* (pp. 113–145). Springer. Retrieved from http://link.springer.com/chapter/10.1007/978-1-4020-5725-0_5
- Briffa, Keith R., Jones, P. D., Schweingruber, F. H., Karlén, W., & Shiyatov, S. G. (1996). Tree-ring variables as proxy-climate indicators: Problems with low-frequency signals. In P. D. Jones, R. S. Bradley, & J. Jouzel (Eds.), *Climatic Variations and Forcing Mechanisms of the Last 2000 Years* (pp. 9–41). Springer Berlin Heidelberg. Retrieved from http://link.springer.com/chapter/10.1007/978-3-642-61113-1_2

- Brodrribb, T. J., & Cochard, H. (2009). Hydraulic Failure Defines the Recovery and Point of Death in Water-Stressed Conifers. *Plant Physiology*, 149(1), 575–584.
<https://doi.org/10.1104/pp.108.129783>
- Brodrribb, T. J., & Holbrook, N. M. (2003). Stomatal Closure during Leaf Dehydration, Correlation with Other Leaf Physiological Traits. *Plant Physiology*, 132(4), 2166–2173.
<https://doi.org/10.1104/pp.103.023879>
- Bunn, A., Korpela, M., Biondi, F., Campelo, F., Mérian, P., Qeadan, F., et al. (2019). dplR: Dendrochronology Program Library in R (Version 1.7.0). Retrieved from <http://cran.r-project.org/web/packages/dplR/index.html>
- Bunn, A. G. (2008). A dendrochronology program library in R (dplR). *Dendrochronologia*, 26(2), 115–124. <https://doi.org/10.1016/j.dendro.2008.01.002>
- Butler, J. J., Kluitenberg, G. J., Whittemore, D. O., Loheide, S. P., Jin, W., Billinger, M. A., & Zhan, X. (2007). A field investigation of phreatophyte-induced fluctuations in the water table. *Water Resources Research*, 43(2), W02404. <https://doi.org/10.1029/2005WR004627>
- Cadol, D., Rathburn, S. L., & Cooper, D. J. (2011). Aerial photographic analysis of channel narrowing and vegetation expansion in Canyon De Chelly National Monument, Arizona, USA, 1935–2004. *River Research and Applications*, 27(7), 841–856. <https://doi.org/10.1002/rra.1399>
- Cailleret, M., Jansen, S., Robert, E. M. R., Desoto, L., Aakala, T., Antos, J. A., et al. (2017). A synthesis of radial growth patterns preceding tree mortality. *Global Change Biology*, 23(4), 1675–1690. <https://doi.org/10.1111/gcb.13535>
- Caskey, S. T., Blaschak, T. S., Wohl, E., Schnackenberg, E., Merritt, D. M., & Dwire, K. A. (2015). Downstream effects of stream flow diversion on channel characteristics and riparian vegetation in the Colorado Rocky Mountains, USA. *Earth Surface Processes and Landforms*, 40(5), 586–598. <https://doi.org/10.1002/esp.3651>
- Chamaillard, S., Fichot, R., Vincent-Barbaroux, C., Bastien, C., Depierreux, C., Dreyer, E., et al. (2011). Variations in bulk leaf carbon isotope discrimination, growth and related leaf traits among three *Populus nigra* L. populations. *Tree Physiology*, 31(10), 1076–1087.
<https://doi.org/10.1093/treephys/tpr089>
- Chin, A., Florsheim, J. L., Wohl, E., & Collins, B. D. (2014). Feedbacks in Human–Landscape Systems. *Environmental Management*, 53(1), 28–41. <https://doi.org/10.1007/s00267-013-0031-y>
- Clow, D. W. (2010). Changes in the Timing of Snowmelt and Streamflow in Colorado: A Response to Recent Warming. *Journal of Climate*, 23(9), 2293–2306.
<https://doi.org/10.1175/2009JCLI2951.1>
- Cook, E. R. (1985). A Time Series Analysis Approach to Tree Ring Standardization. *PhD Dissertation*. Retrieved from <http://arizona.openrepository.com/arizona/handle/10150/188110>

- Cook, E. R., Meko, D. M., Stahle, D. W., & Cleaveland, M. K. (1999). Drought Reconstructions for the Continental United States*. *Journal of Climate*, 12(4), 1145–1162. [https://doi.org/10.1175/1520-0442\(1999\)012<1145:DRFTCU>2.0.CO;2](https://doi.org/10.1175/1520-0442(1999)012<1145:DRFTCU>2.0.CO;2)
- Cook, E. R., Palmer, J. G., Ahmed, M., Woodhouse, C. A., Fenwick, P., Zafar, M. U., et al. (2013). Five Centuries of Upper Indus River Flow from Tree Rings. *Journal of Hydrology*, 486, 365–375. <https://doi.org/10.1016/j.jhydrol.2013.02.004>
- Cooper, D. J., D’Amico, D. R., & Scott, M. L. (2003). Physiological and morphological response patterns of *Populus deltoides* to alluvial groundwater pumping. *Environmental Management*, 31(2), 215–226. <https://doi.org/10.1007/s00267-002-2808-2>
- Covino, T., McGlynn, B., & Mallard, J. (2011). Stream-groundwater exchange and hydrologic turnover at the network scale. *Water Resources Research*, 47(12). <https://doi.org/10.1029/2011WR010942>
- Csank, A. Z., Miller, A. E., Sherriff, R. L., Berg, E. E., & Welker, J. M. (2016). Tree-ring isotopes reveal drought sensitivity in trees killed by spruce beetle outbreaks in south-central Alaska. *Ecological Applications*, 26(7), 2001–2020. <https://doi.org/10.1002/eap.1365>
- Daly, C., Halbleib, M., Smith, J. I., Gibson, W. P., Doggett, M. K., Taylor, G. H., et al. (2008). Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. *International Journal of Climatology*, 28(15), 2031–2064. <https://doi.org/10.1002/joc.1688>
- David, G. C. L., Bledsoe, B. P., Merritt, D. M., & Wohl, E. (2009). The impacts of ski slope development on stream channel morphology in the White River National Forest, Colorado, USA. *Geomorphology*, 103(3), 375–388. <https://doi.org/10.1016/j.geomorph.2008.07.003>
- Dean, D. J., Scott, M. L., Shafroth, P. B., & Schmidt, J. C. (2011). Stratigraphic, sedimentologic, and dendrogeomorphic analyses of rapid floodplain formation along the Rio Grande in Big Bend National Park, Texas. *GSA Bulletin*, 123(9–10), 1908–1925. <https://doi.org/10.1130/B30379.1>
- Díaz-Redondo, M., Marchamalo, M., Egger, G., & Magdaleno, F. (2018). Toward floodplain rejuvenation in the middle Ebro River (Spain): From history to action. *Geomorphology*, 317, 117–127. <https://doi.org/10.1016/j.geomorph.2018.05.014>
- Diehl, R. M., Wilcox, A. C., Merritt, D. M., Perkins, D. W., & Scott, J. A. (2018). Development of an eco-geomorphic modeling framework to evaluate riparian ecosystem response to flow-regime changes. *Ecological Engineering*, 123, 112–126. <https://doi.org/10.1016/j.ecoleng.2018.08.024>
- Dunlap, J. M., Braatne, J. H., Hinckley, T. M., & Stettler, R. F. (1993). Intraspecific variation in photosynthetic traits of *Populus trichocarpa*. *Canadian Journal of Botany*, 71(10), 1304–1311. <https://doi.org/10.1139/b93-156>

- Erskine, W., Keene, A., Bush, R., Cheetham, M., & Chalmers, A. (2012). Influence of riparian vegetation on channel widening and subsequent contraction on a sand-bed stream since European settlement: Widden Brook, Australia. *Geomorphology*, 147–148, 102–114. <https://doi.org/10.1016/j.geomorph.2011.07.030>
- Everitt, B. L. (1968). Use of the cottonwood in an investigation of the recent history of a flood plain. *American Journal of Science*, 266(6), 417–439. <https://doi.org/10.2475/ajs.266.6.417>
- Farquhar, G., O’Leary, M., & Berry, J. (1982). On the Relationship Between Carbon Isotope Discrimination and the Intercellular Carbon Dioxide Concentration in Leaves. *Functional Plant Biology*, 9(2), 121. <https://doi.org/10.1071/PP9820121>
- Ficklin, D. L., & Novick, K. A. (2017). Historic and projected changes in vapor pressure deficit suggest a continental-scale drying of the United States atmosphere. *Journal of Geophysical Research: Atmospheres*, 122(4), 2061–2079. <https://doi.org/10.1002/2016JD025855>
- Frank, D., Reichstein, M., Bahn, M., Thonicke, K., Frank, D., Mahecha, M. D., et al. (2015). Effects of climate extremes on the terrestrial carbon cycle: concepts, processes and potential future impacts. *Global Change Biology*, 21(8), 2861–2880. <https://doi.org/10.1111/gcb.12916>
- Friedman, J. M., Auble, G. T., Shafroth, P. B., Scott, M. L., Merigliano, M. F., Freehling, M. D., & Griffin, E. R. (2005). Dominance of non-native riparian trees in western USA. *Biological Invasions*, 7(4), 747–751. <http://dx.doi.org/10.1007/s10530-004-5849-z>
- Friedman, J. M., Stricker, C. A., Csank, A. Z., & Zhou, H. (2019). Effects of age and environment on stable carbon isotope ratios in tree rings of riparian *Populus*. *Palaeogeography, Palaeoclimatology, Palaeoecology*, 524, 25–32. <https://doi.org/10.1016/j.palaeo.2019.03.022>
- Gaudinski, J. B., Dawson, T. E., Quideau, S., Schuur, E. A. G., Roden, J. S., Trumbore, S. E., et al. (2005). Comparative Analysis of Cellulose Preparation Techniques for Use with ¹³C, ¹⁴C, and ¹⁸O Isotopic Measurements. *Analytical Chemistry*, 77(22), 7212–7224. <https://doi.org/10.1021/ac050548u>
- Gleason, S. M., Blackman, C. J., Cook, A. M., Laws, C. A., & Westoby, M. (2014). Whole-plant capacitance, embolism resistance and slow transpiration rates all contribute to longer desiccation times in woody angiosperms from arid and wet habitats. *Tree Physiology*, 34(3), 275–284. <https://doi.org/10.1093/treephys/tpu001>
- Gonzalez, M. A. (2001). Recent formation of arroyos in the Little Missouri Badlands of southwestern North Dakota. *Geomorphology*, 38(1–2), 63–84. [https://doi.org/10.1016/S0169-555X\(00\)00070-2](https://doi.org/10.1016/S0169-555X(00)00070-2)
- Gray, S. T., Graumlich, L. J., & Betancourt, J. L. (2007). Annual precipitation in the Yellowstone National Park region since AD 1173. *Quaternary Research*, 68(1), 18–27.

- Grismer, M. E. (2018). Putah Creek hydrology affecting riparian cottonwood and willow tree survival. *Environmental Monitoring and Assessment*, 190(8). <https://doi.org/10.1007/s10661-018-6841-x>
- Grissino-Mayer, H. D. (2001). Evaluating Crossdating Accuracy: A Manual and Tutorial for the Computer Program COFECHA. *Tree-Ring Research*, 57(2), 205–221.
- Gurnell, A. M., Bertoldi, W., & Corenblit, D. (2012). Changing river channels: The roles of hydrological processes, plants and pioneer fluvial landforms in humid temperate, mixed load, gravel bed rivers. *Earth-Science Reviews*, 111(1–2), 129–141. <https://doi.org/10.1016/j.earscirev.2011.11.005>
- Hajdukiewicz, H., & Wyzga, B. (2019). Aerial photo-based analysis of the hydromorphological changes of a mountain river over the last six decades: The Czarny Dunajec, Polish Carpathians. *Science of The Total Environment*, 648, 1598–1613. <https://doi.org/10.1016/j.scitotenv.2018.08.234>
- Harman, C. J., Ward, A. S., & Ball, A. (2016). How does reach-scale stream-hyporheic transport vary with discharge? Insights from rSAS analysis of sequential tracer injections in a headwater mountain stream. *Water Resources Research*, 52(9), 7130–7150. <https://doi.org/10.1002/2016WR018832>
- Holloway, J. V., Rillig, M. C., & Gurnell, A. M. (2017). Underground riparian wood: Buried stem and coarse root structures of Black Poplar (*Populus nigra* L.). *Geomorphology*, 279, 188–198. <https://doi.org/10.1016/j.geomorph.2016.08.002>
- Holmes, R. L. (1983). Computer-assisted quality control in tree-ring dating and measurement. *Tree-Ring Bulletin*, 43, 69–78.
- Johnson, T. C., Slater, L. D., Ntarlagiannis, D., Day-Lewis, F. D., & Elwaseif, M. (2012). Monitoring groundwater-surface water interaction using time-series and time-frequency analysis of transient three-dimensional electrical resistivity changes. *Water Resources Research*, 48(7), 13. <https://doi.org/10.1029/2012WR011893>
- Jump, A. S., Ruiz-Benito, P., Greenwood, S., Allen, C. D., Kitzberger, T., Fensham, R., et al. (2017). Structural overshoot of tree growth with climate variability and the global spectrum of drought-induced forest dieback. *Global Change Biology*, 23(9), 3742–3757. <https://doi.org/10.1111/gcb.13636>
- Kaluthota, S., Pearce, D. W., Evans, L. M., Letts, M. G., Whitham, T. G., & Rood, S. B. (2015). Higher photosynthetic capacity from higher latitude: foliar characteristics and gas exchange of southern, central and northern populations of *Populus angustifolia*. *Tree Physiology*, 35(9), 936–948. <https://doi.org/10.1093/treephys/tpv069>
- Kamp, K. V., Rigge, M., Troelstrup, N. H., Smart, A. J., & Wylie, B. (2013). Detecting Channel Riparian Vegetation Response to Best-Management-Practices Implementation in Ephemeral

- Streams With the Use of Spot High-Resolution Visible Imagery. *Rangeland Ecology & Management*, 66(1), 63–70.
- Kampf, S. K., & Lefsky, M. A. (2016). Transition of dominant peak flow source from snowmelt to rainfall along the Colorado Front Range: Historical patterns, trends, and lessons from the 2013 Colorado Front Range floods. *Water Resources Research*, n/a-n/a. <https://doi.org/10.1002/2015WR017784>
- Kattel, G. R. (2019). State of future water regimes in the world's river basins: balancing the water between society and nature. *Critical Reviews in Environmental Science and Technology*, 49(12), 1107–1133. <https://doi.org/10.1080/10643389.2019.1579621>
- Lane, E. W. (1954). *The Importance of Fluvial Morphology in Hydraulic Engineering*. U.S. Department of the Interior, Bureau of Reclamation, Commissioner's Office.
- Latli, A., Michel, L. N., Lepoint, G., & Kestemont, P. (2019). River habitat homogenisation enhances trophic competition and promotes individual specialisation among young of the year fish. *Freshwater Biology*, 64(3), 520–531. <https://doi.org/10.1111/fwb.13239>
- Leblanc, S. G., Chen, J. M., & Kwong, M. (2002). *Tracing radiation and Architecture of Canopies – TRAC Manual Version 2.1*.
- Leffler, A. J., England, L. E., & Naito, J. (2000). Vulnerability of Fremont cottonwood (*Populus fremontii* Wats.) individuals to xylem cavitation. *Western North American Naturalist*, 60(2), 204–210.
- Liébault, F., Clément, P., Piégay, H., Rogers, C. F., Kondolf, G. M., & Landon, N. (2002). Contemporary channel changes in the Eygues basin, southern French Prealps: the relationship of subbasin variability to watershed characteristics. *Geomorphology*, 45(1), 53–66. [https://doi.org/10.1016/S0169-555X\(01\)00189-1](https://doi.org/10.1016/S0169-555X(01)00189-1)
- Liro, M. (2019). Dam reservoir backwater as a field-scale laboratory of human-induced changes in river biogeomorphology: A review focused on gravel-bed rivers. *Science of The Total Environment*, 651, 2899–2912. <https://doi.org/10.1016/j.scitotenv.2018.10.138>
- Lytle, D. A., & Poff, N. L. (2004). Adaptation to natural flow regimes. *Trends in Ecology & Evolution*, 19(2), 94–100. <https://doi.org/10.1016/j.tree.2003.10.002>
- Mackin, J. H. (1948). Concept of the graded river. *Geological Society of America Bulletin*, 59(5), 463–512.
- Manners, R. B., Wilcox, A. C., Kui, L., Lightbody, A. F., Stella, J. C., & Sklar, L. S. (2015). When do plants modify fluvial processes? Plant-hydraulic interactions under variable flow and sediment supply rates. *Journal of Geophysical Research: Earth Surface*, 120(2), 325–345. <https://doi.org/10.1002/2014JF003265>

- McCarroll, D., & Loader, N. J. (2004). Stable isotopes in tree rings. *Quaternary Science Reviews*, 23(7–8), 771–801. <https://doi.org/10.1016/j.quascirev.2003.06.017>
- McDowell, N., Pockman, W. T., Allen, C. D., Breshears, D. D., Cobb, N., Kolb, T., et al. (2008). Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? *New Phytologist*, 178(4), 719–739. <https://doi.org/10.1111/j.1469-8137.2008.02436.x>
- McDowell, N. G., Beerling, D. J., Breshears, D. D., Fisher, R. A., Raffa, K. F., & Stitt, M. (2011). The interdependence of mechanisms underlying climate-driven vegetation mortality. *Trends in Ecology & Evolution*, 26(10), 523–532. <https://doi.org/10.1016/j.tree.2011.06.003>
- McGrew, A. J., & Miller, E. L. (1995). *Geologic map of Kiou Spring and Garrison 7.5' quadrangles, White Pine County, Nevada and Millard County, Utah* (USGS Numbered Series No. 95–10). Retrieved from <http://pubs.er.usgs.gov/publication/ofr9510>
- McKay, S. F., & King, A. J. (2006). Potential ecological effects of water extraction in small, unregulated streams. *River Research and Applications*, 22(9), 1023–1037. <https://doi.org/10.1002/rra.958>
- Meko, D. M., Touchan, R., & Anchukaitis, K. J. (2011). Seascorr: A MATLAB program for identifying the seasonal climate signal in an annual tree-ring time series. *Computers & Geosciences*, 37(9), 1234–1241. <https://doi.org/10.1016/j.cageo.2011.01.013>
- Meko, David M., Woodhouse, C. A., Baisan, C. A., Knight, T., Lukas, J. J., Hughes, M. K., & Salzer, M. W. (2007). Medieval drought in the upper Colorado River Basin. *Geophysical Research Letters*, 34(10). <https://doi.org/10.1029/2007GL029988>
- Meko, David M., Friedman, J. M., Touchan, R., Edmondson, J. R., Griffin, E. R., & Scott, J. A. (2015). Alternative standardization approaches to improving streamflow reconstructions with ring-width indices of riparian trees. *The Holocene*, 25(7), 1093–1101. <https://doi.org/10.1177/0959683615580181>
- Merigliano, M. F., Friedman, J. M., & Scott, M. L. (2013). 12.10 Tree-Ring Records of Variation in Flow and Channel Geometry. In J. F. Shroder (Ed.), *Treatise on Geomorphology* (pp. 145–164). San Diego: Academic Press. Retrieved from <http://www.sciencedirect.com/science/article/pii/B9780123747396003195>
- Merritt, D. M., & Cooper, D. J. (2000). Riparian vegetation and channel change in response to river regulation: a comparative study of regulated and unregulated streams in the Green River Basin, USA. *Regulated Rivers: Research & Management*, 16(6), 543–564. [https://doi.org/10.1002/1099-1646\(200011/12\)16:6<543::AID-RRR590>3.0.CO;2-N](https://doi.org/10.1002/1099-1646(200011/12)16:6<543::AID-RRR590>3.0.CO;2-N)
- Metzger, T. L., Pizzuto, J. E., Trampush, S. M., Friedman, J. M., & Schook, D. M. (2019). Event-Scale Floodplain Accretion Revealed Through Tree-Ring Analysis of Buried Plains

- Cottonwoods, Powder River, MT, USA. *Earth Surface Processes and Landforms*, 0. <https://doi.org/10.1002/esp.4734>
- Monahan, W. B., & Fisichelli, N. A. (2014). Climate Exposure of US National Parks in a New Era of Change. *PLOS ONE*, 9(7), e101302. <https://doi.org/10.1371/journal.pone.0101302>
- Montgomery, D. R., & Buffington, J. M. (1997). Channel-reach morphology in mountain drainage basins. *Geological Society of America Bulletin*, 109(5), 596–611. [https://doi.org/10.1130/0016-7606\(1997\)109<0596:CRMIMD>2.3.CO;2](https://doi.org/10.1130/0016-7606(1997)109<0596:CRMIMD>2.3.CO;2)
- Naiman, R. J., Decamps, H., & McClain, M. E. (2010). *Riparia: Ecology, Conservation, and Management of Streamside Communities*. Academic Press.
- Nakagawa, S., & Cuthill, I. C. (2007). Effect size, confidence interval and statistical significance: a practical guide for biologists. *Biological Reviews of the Cambridge Philosophical Society*, 82(4), 591–605. <https://doi.org/10.1111/j.1469-185X.2007.00027.x>
- Nakano, S., Miyasaka, H., & Kuhara, N. (1999). Terrestrial–aquatic linkages: riparian arthropod inputs alter trophic cascades in a stream food web. *Ecology*, 80(7), 2435–2441.
- Nilsson, C., & Berggren, K. (2000). Alterations of Riparian Ecosystems Caused by River Regulation. *BioScience*, 50(9), 783. [https://doi.org/10.1641/0006-3568\(2000\)050\[0783:AORECB\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2000)050[0783:AORECB]2.0.CO;2)
- NOAA. (2018). Retrieved from <https://www.ncdc.noaa.gov/>
- Osterkamp, W. R., & Hedman, E. R. (1982). *Perennial-streamflow characteristics related to channel geometry and sediment in Missouri River basin* (USGS Numbered Series No. 1242). Retrieved from <http://pubs.er.usgs.gov/publication/pp1242>
- Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I.-C., et al. (2017). Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science*, 355(6332), eaai9214. <https://doi.org/10.1126/science.aai9214>
- Perkin, J. S., Gido, K. B., Cooper, A. R., Turner, T. F., Osborne, M. J., Johnson, E. R., & Mayes, K. B. (2015). Fragmentation and dewatering transform Great Plains stream fish communities. *Ecological Monographs*, 85(1), 73–92. <https://doi.org/10.1890/14-0121.1>
- Poepl, R. E., Keesstra, S. D., & Maroulis, J. (2017). A conceptual connectivity framework for understanding geomorphic change in human-impacted fluvial systems. *Geomorphology*, 277, 237–250. <https://doi.org/10.1016/j.geomorph.2016.07.033>
- Poff, N. L., & Zimmerman, J. K. H. (2010). Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology*, 55(1), 194–205. <https://doi.org/10.1111/j.1365-2427.2009.02272.x>

- Pollen, N., & Simon, A. (2005). Estimating the mechanical effects of riparian vegetation on stream bank stability using a fiber bundle model. *Water Resources Research*, 41(7). <https://doi.org/10.1029/2004WR003801>
- PRISM Climate Group. (2018). PRISM Climate Group. Retrieved from <http://www.prism.oregonstate.edu>
- Prudic, D. E., Sweetkind, D. S., Jackson, T. R., Dotson, K. E., Plume, R. W., Hatch, C. E., & Halford, K. J. (2015). *Evaluating connection of aquifers to springs and streams, Great Basin National Park and vicinity, Nevada* (USGS Numbered Series No. 1819). Reston, VA: U.S. Geological Survey. Retrieved from <http://pubs.er.usgs.gov/publication/pp1819>
- Reynolds, L. V., Shafroth, P. B., & House, P. K. (2014). Abandoned Floodplain Plant Communities Along a Regulated Dryland River. *River Research and Applications*, 30(9), 1084–1098. <https://doi.org/10.1002/rra.2708>
- Rood, S.B., Patiño, S., Coombs, K., & Tyree, M. T. (2000). Branch sacrifice: cavitation-associated drought adaptation of riparian cottonwoods. *Trees*, 14(5), 0248–0257. <https://doi.org/10.1007/s004680050010>
- Rood, Stewart B., & Heinze-Milne, S. (1989). Abrupt downstream forest decline following river damming in southern Alberta. *Canadian Journal of Botany*, 67(6), 1744–1749. <https://doi.org/10.1139/b89-221>
- Rood, Stewart B., & Mahoney, J. M. (1990). Collapse of riparian poplar forests downstream from dams in western prairies: probable causes and prospects for mitigation. *Environmental Management*, 14(4), 451–464.
- Rood, Stewart B., Gourley, C. R., Ammon, E. M., Heki, L. G., Klotz, J. R., Morrison, M. L., et al. (2003). Flows for Floodplain Forests: A Successful Riparian Restoration. *BioScience*, 53(7), 647–656. [https://doi.org/10.1641/0006-3568\(2003\)053\[0647:FFFFAS\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2003)053[0647:FFFFAS]2.0.CO;2)
- Rood, Stewart B., Bigelow, S. G., & Hall, A. A. (2011). Root architecture of riparian trees: river cut-banks provide natural hydraulic excavation, revealing that cottonwoods are facultative phreatophytes. *Trees*, 25(5), 907–917. <https://doi.org/10.1007/s00468-011-0565-7>
- Rood, Stewart B., Ball, D. J., Gill, K. M., Kaluthota, S., Letts, M. G., & Pearce, D. W. (2013). Hydrologic linkages between a climate oscillation, river flows, growth, and wood $\Delta^{13}\text{C}$ of male and female cottonwood trees: Hydrological linkages for riparian cottonwoods. *Plant, Cell & Environment*, 36(5), 984–993. <https://doi.org/10.1111/pce.12031>
- Roux, C., Alber, A., Bertrand, M., Vaudor, L., & Piégay, H. (2015). “FluvialCorridor”: A new ArcGIS toolbox package for multiscale riverscape exploration. *Geomorphology*, 242, 29–37. <https://doi.org/10.1016/j.geomorph.2014.04.018>

- Ryan, S. (1997). Morphologic Response of Subalpine Streams to Transbasin Flow Diversion. *Journal of the American Water Resources Association*, 33(4), 839–854. <https://doi.org/10.1111/j.1752-1688.1997.tb04109.x>
- Sabo, J. L., & Power, M. E. (2002). Numerical response of lizards to aquatic insects and short-term consequences for terrestrial prey. *Ecology*, 83(11), 3023–3036.
- Schifman, L. A., Stella, J. C., Volk, T. A., & Teece, M. A. (2012). Carbon isotope variation in shrub willow (*Salix* spp.) ring-wood as an indicator of long-term water status, growth and survival. *Biomass and Bioenergy*, 36, 316–326. <https://doi.org/10.1016/j.biombioe.2011.10.042>
- Schleser, G. H., Anhuf, D., Helle, G., & Vos, H. (2015). A remarkable relationship of the stable carbon isotopic compositions of wood and cellulose in tree-rings of the tropical species *Cariniana micrantha* (Ducke) from Brazil. *Chemical Geology*, 401, 59–66. <https://doi.org/10.1016/j.chemgeo.2015.02.014>
- Schollaen, K., Heinrich, I., Neuwirth, B., Krusic, P. J., D'Arrigo, R. D., Karyanto, O., & Helle, G. (2013). Multiple tree-ring chronologies (ring width, $\delta^{13}\text{C}$ and $\delta^{18}\text{O}$) reveal dry and rainy season signals of rainfall in Indonesia. *Quaternary Science Reviews*, 73, 170–181. <https://doi.org/10.1016/j.quascirev.2013.05.018>
- Schomaker, M. E., Zarnoch, S. J., Bechtold, W. A., Latelle, D. J., Burkman, W. G., & Cox, S. M. (2007). Crown-condition classification: a guide to data collection and analysis. *Gen. Tech. Rep. SRS-102*. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 78 p., 102. Retrieved from <https://www.fs.usda.gov/treesearch/pubs/27730>
- Schook, D. M., Carlson, E. A., Sholtes, J. S., & Cooper, D. J. (2016). Effects of Moderate and Extreme Flow Regulation on *Populus* Growth along the Green and Yampa Rivers, Colorado and Utah. *River Research and Applications*. <https://doi.org/10.1002/rra.3020>
- Schook, D. M., Friedman, J. M., & Rathburn, S. L. (2016). Flow reconstructions in the Upper Missouri River Basin using riparian tree rings. *Water Resources Research*, 52, 8159–8173. <https://doi.org/10.1002/2016WR018845>
- Schook, D. M., Rathburn, S. L., Friedman, J. M., & Wolf, J. M. (2017). A 184-year record of river meander migration from tree rings, aerial imagery, and cross sections. *Geomorphology*, 293, 227–239. <https://doi.org/10.1016/j.geomorph.2017.06.001>
- Schulte, P. J., Hinckley, T. M., & Stettler, R. F. (1987). Stomatal responses of *Populus* to leaf water potential. *Canadian Journal of Botany*, 65(2), 255–260. <https://doi.org/10.1139/b87-036>
- Schweingruber, F. H. (2012). *Tree Rings: Basics and Applications of Dendrochronology*. Springer Science & Business Media.

- Scoffoni, C., Rawls, M., McKown, A., Cochard, H., & Sack, L. (2011). Decline of leaf hydraulic conductance with dehydration: relationship to leaf size and venation architecture. *Plant Physiology*, 156, 832–843.
- Scott, M. L., Auble, G. T., & Friedman, J. M. (1997). Flood Dependency of Cottonwood Establishment Along the Missouri River, Montana, USA. *Ecological Applications*, 7(2), 677–690. [https://doi.org/10.1890/1051-0761\(1997\)007\[0677:FDOCEA\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1997)007[0677:FDOCEA]2.0.CO;2)
- Scott, M. L., Shafroth, P. B., & Auble, G. T. (1999). Responses of Riparian Cottonwoods to Alluvial Water Table Declines. *Environmental Management*, 23(3), 347–358. <https://doi.org/10.1007/s002679900191>
- Scott, M. L., Lines, G. C., & Auble, G. T. (2000). Channel incision and patterns of cottonwood stress and mortality along the Mojave River, California. *Journal of Arid Environments*, 44(4), 399–414. <https://doi.org/10.1006/jare.1999.0614>
- Secchi, F., & Zwieniecki, M. A. (2010). Patterns of PIP gene expression in *Populus trichocarpa* during recovery from xylem embolism suggest a major role for the PIP1 aquaporin subfamily as moderators of refilling process. *Plant, Cell and Environment*, 33(8), 1285–1297. <https://doi.org/10.1111/j.1365-3040.2010.02147.x>
- Shafroth, P. B., Stromberg, J. C., & Patten, D. T. (2000). Woody riparian vegetation response to different alluvial water table regimes. *Western North American Naturalist*, 60(1), 66–76.
- Shafroth, P. B., Stromberg, J. C., & Patten, D. T. (2002). Riparian vegetation response to altered disturbance and stress regimes. *Ecological Applications*, 12(1), 107–123. [https://doi.org/10.1890/1051-0761\(2002\)012\[0107:RVRTAD\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2002)012[0107:RVRTAD]2.0.CO;2)
- Singer, M. B., Stella, J. C., Dufour, S., Piégay, H., Wilson, R. J. S., & Johnstone, L. (2013). Contrasting water-uptake and growth responses to drought in co-occurring riparian tree species. *Ecohydrology*, 6(3), 402–412. <https://doi.org/10.1002/eco.1283>
- Smith, S. D., Wellington, A. B., Nachlinger, J. L., & Fox, C. A. (1991). Functional Responses of Riparian Vegetation to Streamflow Diversion in the Eastern Sierra Nevada. *Ecological Applications*, 1(1), 89–97. <https://doi.org/10.2307/1941850>
- Sparacino, M. S., Rathburn, S. L., Covino, T. P., Singha, K., & Ronayne, M. J. (2019). Form-based river restoration decreases wetland hyporheic exchange: Lessons learned from the Upper Colorado River: Form-based river restoration decreases wetland hyporheic exchange. *Earth Surface Processes and Landforms*, 44(1), 191–203. <https://doi.org/10.1002/esp.4525>
- Sperry, J. S., & Love, D. M. (2015). What plant hydraulics can tell us about responses to climate-change droughts. *New Phytologist*, 207(1), 14–27. <https://doi.org/10.1111/nph.13354>
- St. George, S. (2014). An overview of tree-ring width records across the Northern Hemisphere. *Quaternary Science Reviews*, 95, 132–150. <https://doi.org/10.1016/j.quascirev.2014.04.029>

- Stahle, D. W., Cook, E. R., Cleaveland, M. K., Therrell, M. D., Meko, D. M., Grissino-Mayer, H. D., et al. (2000). Tree-ring data document 16th century megadrought over North America. *Eos, Transactions American Geophysical Union*, 81(12), 121–125. <https://doi.org/10.1029/00EO00076>
- Stokes, M. A., & Smiley, T. L. (1968). *An introduction to tree-ring dating*. University of Chicago Press.
- Stromberg, J. C. (2001). Restoration of riparian vegetation in the south-western United States: importance of flow regimes and fluvial dynamism. *Journal of Arid Environments*, 49(1), 17–34. <https://doi.org/10.1006/jare.2001.0833>
- Stromberg, J. C., & Merritt, D. M. (2016). Riparian plant guilds of ephemeral, intermittent and perennial rivers. *Freshwater Biology*, 61(8), 1259–1275. <https://doi.org/10.1111/fwb.12686>
- Stromberg, J. C., & Patten, D. T. (1990). Riparian vegetation instream flow requirements: A case study from a diverted stream in the Eastern Sierra Nevada, California, USA. *Environmental Management*, 14(2), 185–194. <https://doi.org/10.1007/BF02394035>
- Tai, X., Mackay, D. S., Sperry, J. S., Brooks, P., Anderegg, W. R. L., Flanagan, L. B., et al. (2018). Distributed Plant Hydraulic and Hydrological Modeling to Understand the Susceptibility of Riparian Woodland Trees to Drought-Induced Mortality. *Water Resources Research*, 54(7), 4901–4915.
- Treydte, K., Frank, D., Esper, J., Andreu, L., Bednarz, Z., Berninger, F., et al. (2007). Signal strength and climate calibration of a European tree-ring isotope network. *Geophysical Research Letters*, 34(24). <https://doi.org/10.1029/2007GL031106>
- Tyree, M. T., Kolb, K. J., Rood, S. B., & Patiño, S. (1994). Vulnerability to drought-induced cavitation of riparian cottonwoods in Alberta: a possible factor in the decline of the ecosystem? *Tree Physiology*, 14(5), 455–466. <https://doi.org/10.1093/treephys/14.5.455>
- Udall, B., & Overpeck, J. (2017). The twenty-first century Colorado River hot drought and implications for the future. *Water Resources Research*, 53(3), 2404–2418. <https://doi.org/10.1002/2016WR019638>
- Van Hoesen, J. G. (2002). *Great Basin National Park Technical Assistance Proposal – Hydrology and Geohydrology of Snake Creek* (Final Report) (p. 23). Las Vegas, NV: University of Las Vegas, Nevada.
- Warren, C. R., McGrath, J. F., & Adams, M. A. (2001). Water availability and carbon isotope discrimination in conifers. *Oecologia*, 127(4), 476–486. <https://doi.org/10.1007/s004420000609>
- Williams, A. P., Allen, C. D., Macalady, A. K., Griffin, D., Woodhouse, C. A., Meko, D. M., et al. (2013). Temperature as a potent driver of regional forest drought stress and tree mortality. *Nature Climate Change*, 3(3), 292–297. <https://doi.org/10.1038/nclimate1693>

- Williams, C. A., & Cooper, D. J. (2005). Mechanisms of riparian cottonwood decline along regulated rivers. *Ecosystems*, 8(4), 382–395. <https://doi.org/10.1007/s10021-003-0072-9>
- Willms, C. R., Pearce, D. W., & Rood, S. B. (2006). Growth of riparian cottonwoods: a developmental pattern and the influence of geomorphic context. *Trees*, 20(2), 210–218. <https://doi.org/10.1007/s00468-005-0027-1>
- Wilson, A. T., & Grinsted, M. J. (1977). $\delta^{12}\text{C}$ / $\delta^{13}\text{C}$ in cellulose and lignin as palaeothermometers. *Nature*, 265(5590), 133–135. <https://doi.org/10.1038/265133a0>
- Wohl, E. (2014). *Rivers in the Landscape: Science and Management*. Wiley. Retrieved from <http://books.google.com/books?id=py4KAwAAQBAJ>
- Wohl, Ellen. (2001). *Virtual Rivers: Lessons from the Mountain Rivers of the Colorado Front Range*. New Haven; London: Yale University Press.
- Wohl, Ellen, Bledsoe, B. P., Jacobson, R. B., Poff, N. L., Rathburn, S. L., Walters, D. M., & Wilcox, A. C. (2015). The Natural Sediment Regime in Rivers: Broadening the Foundation for Ecosystem Management. *BioScience*, biv002. <https://doi.org/10.1093/biosci/biv002>
- Wyckoff, P. H., & Clark, J. S. (2000). Predicting tree mortality from diameter growth: a comparison of maximum likelihood and Bayesian approaches. *Canadian Journal of Forest Research*, 30(1), 156–167. <https://doi.org/10.1139/x99-198>

16. Appendix

16.1 Tree ring chronologies

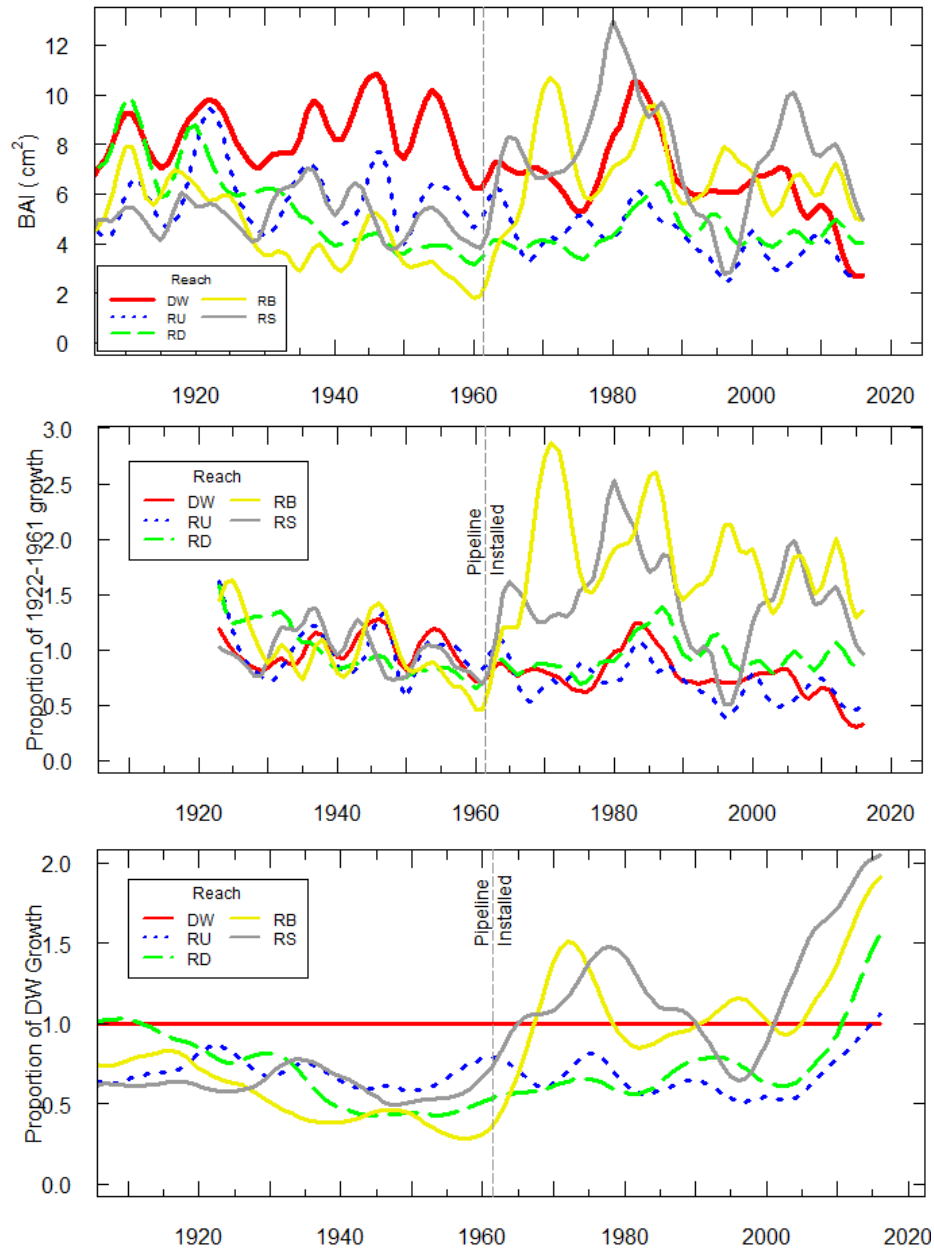


Figure 16.1. Top panel shows smoothed annual values of tree ring BAI for the five study reaches. Note that growth in DW was highest for most of the pre-pipeline period. Middle panel shows smoothed annual values of BAI relative to the 40-year (1922–1961) pre-pipeline period, where growth at all reaches is standardized to average 1. Bottom panel shows smoothed annual values of BAI per reach relative to that at DW for each year. Bottom panel highlights that growth was highest at DW for nearly the entire pre-pipeline period; immediately after pipeline installation RS and RB growth became larger relative to DW, and a similar shift occurred at RU and RD in the 2010s.

16.2 Aerial photos

Table 16.1. The multiscale segmentation algorithm was used at two different settings to identify coarse- and fine-scale objects in the object-based image analysis (OBIA). The “scale parameter” and “compactness” variables influenced the size of objects, and larger numbers created larger objects. The “shape” setting influenced how color was handled, and higher values has less influence from color.

Setting	Coarse-scale	Fine-scale
Scale Parameter	125	20
Compactness	0.5	0.2
Shape	0.2	0.2

Table 16.2. Expanded Table Table 8.4. Linear models of mean NDVI value and percent live cottonwood canopy per reach. Interaction models identified if the slope of the lines was steeper in the Dewatered reach. Included are mean NDVI values, t-statistics for the interaction models, and the p-value of between-reach mean NDVI values.

Year	Reference Reaches			Dewatered Reach			Statistics		
	NDVI mean	Slope (NDVI/Canopy%)	R ²	NDVI mean	Slope (NDVI/Canopy%)	R ²	Interaction models: p-value differentiating % canopy slopes	t-stat_for between reach NDVI	p-val_for between-reach means
1985	142.89	0.69	0.32	129.03	-0.28	0.23	0.248	-6.54	<0.001
2010	161.65	0.15	0.27	155.66	1.26	0.97	0.002	-11.65	<0.001
2013	192.29	0.49	0.82	178.53	1.71	0.95	< 0.001	-23.91	<0.001
2015	165.58	1.07	0.64	150.38	3.38	0.96	0.068	-23.53	<0.001
2017	187.14	1.09	0.63	168.98	3.91	0.96	0.038	-17.86	<0.001

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

NPS 148/167853, April 2020

National Park Service
U.S. Department of the Interior



[Natural Resource Stewardship and Science](#)

1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525