Upper Deschutes Watershed Council
Technical Report

2015 Whychus Creek Monitoring Report

Mork, L. and Houston, R. Editors

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Symbols and Abbreviations

BLM	Bureau of Land Management
CTWS	Confederated Tribes of the Warm Springs Reservation
DRC	Deschutes River Conservancy
FERC	Federal Energy Regulatory Commission
NIST	National Institute of Standards and Technology
NOAA	National Oceanic and Atmospheric Admistration
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
OWEB	Oregon Watershed Enhancement Board
OWRD	Oregon Water Resources Department
PGE	Portland General Electric
TSID	Three Sisters Irrigation District
UDWC	Upper Deschutes Watershed Council
USFS	United States Forest Service
USGS	United States Geological Survey

7DMAX	Seven day moving average maximum temperature
°C	Degree Celsius
cfs	Cubic feet per second
df	Degrees of freedom
DO	Dissolved oxygen
°F	Fahrenheit
mg/L	Milligrams per liter
OAR	Oregon Administrative Rules
QA/QC	Quality assurance / quality control
PI	Prediction Interval
S	Standard distance from regression line
TMDL	Total Maximum Daily Load

Restoration Effectiveness Monitoring in Whychus Creek

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Introduction

Local, federal, state, and private agencies and organizations have coalesced around the reintroduction of steelhead into Whychus Creek. The creek, a tributary to Oregon's Deschutes River, was historically one of the most important steelhead spawning streams in the upper Deschutes Basin (Nehlsen 1995). The construction of the Pelton Round Butte dam complex on the Deschutes River in the 1960s eliminated anadromous runs in Whychus Creek.

Fisheries managers agreed to restore fish passage at the Pelton Round Butte dam complex and reintroduce anadromous fish above the facility as part of a hydroelectric relicensing agreement signed in 2005 (FERC 2005). A group of non-profits, public agencies, and private actors had informally cooperated since the mid 1990s to restore habitat conditions in Whychus Creek. The selection of the creek as a focal area for reintroduction catalyzed existing restoration efforts, drawing state and regional restoration investors to the region.

As restoration investments and commitments increased, restoration partners recognized a need to formalize their relationships. The Bonneville Environmental Foundation led the development of the Upper Deschutes Model Watershed in 2006 to foster collaboration between organizations committed to restoring aquatic and riparian habitat in the upper Deschutes Basin. This program, led by the Upper Deschutes Watershed Council (UDWC), provides a nucleus for coordinated restoration in Whychus Creek.

Restoration funders have increasingly looked to quantify the ecological outcomes of their investments. Habitat improvement projects *should* lead to more resilient fish populations. Fish passage projects *should* lead to increased spawning upstream of historic barriers. Stream flow restoration *should* lead to cooler stream temperatures. The lack of monitoring associated with river restoration (Bash and Ryan 2002, O'Donnell and Galat 2008, Souchon *et al* 2008) has made it difficult to quantify these outcomes, let alone document cause-and-effect relationships between specific actions and ecological outcomes.

So, why are so few restoration practitioners monitoring? A survey of 85 restoration project managers in Washington identified limited resources as the primary obstacle to restoration project evaluation (Bash and Ryan 2002). Experiences in the Deschutes Basin suggest that the traditional project-based funding model grossly underfunds monitoring. Project-based restoration funding available through grants typically offers little, if any, opportunity for long-term monitoring. Grants are short-term, focused on immediate results and driven by budget cycles rather than ecological processes. This funding model leads restoration practitioners to focus on implementing projects instead of monitoring outcomes. The Upper Deschutes Model Watershed approach to monitoring restoration effectiveness in Whychus Creek acknowledges these limitations and seeks to leverage limited resources to improve monitoring. The UDWC developed a monitoring approach for Whychus Creek that focuses on tracking the status and trends of select physical and biological indicators. These indicators represent conditions in the creek, prior to and following the implementation of a suite of restoration projects begun in 2009.

In an ideal active adaptive management watershed restoration scenario, restoration practitioners would hypothesize about how individual restoration activities would affect stream structures and functions or lead to responses in target species. Practitioners would then design each restoration activity as an experiment and evaluate their hypotheses using controls, statistical tools and other standard experimental practices.

While this scenario may appear to be ideal, it is not possible in Whychus Creek for three reasons. First, the multiple restoration actions occurring simultaneously along the creek make it difficult to verify cause and effect relationships between specific actions and changes in physical and biological conditions. Second, the multiple agencies and organizations managing and restoring Whychus Creek work under different mandates set by local, state or federal regulations, community interests or other factors. These different mandates make it impractical to establish controls for the rigorous experimental designs necessary for validation monitoring. Finally, there are very limited resources available for monitoring in Whychus Creek. Therefore, from a practical standpoint, any monitoring must be accomplished as efficiently as possible by using existing data. The reliance on existing data inherently limits the types of analyses and the conclusions that can be developed.

The monitoring approach selected by UDWC focuses on tracking the status and trends of key physical and biological indicators in Whychus Creek. We selected these indicators based on a conceptual model of factors limiting salmonid production in the creek (Figure 1), on the premise that ongoing restoration actions will ameliorate the limiting factors identified in the conceptual model and that selected indicators will respond to changes in these limiting factors. This approach will not test cause and effect relationships between restoration actions and changes in selected indicators. It will, however, demonstrate whether these indicators have moved closer to desired conditions. We drew indicators from seven broad categories: stream flow, water quality, habitat quality, stream connectivity, fish entrainment, macroinvertebrates, and fish populations. Each individual section of the Whychus Creek Monitoring Report assesses indicators in one of these categories.

Figure 1.
This conceptual model illustrates the expected influences on each life stage of resident and anadromous salmonids in Whychus Creek. The UDWC expects that the ongoing restoration actions will affect the limiting factors identified in the conceptual model, as measured by the specified monitoring indicators.

Study Area

Whychus Creek originates in the Cascade Mountains near Sisters, OR. The creek's watershed encompasses approximately 162,000 acres and 40 stream miles in Deschutes and Jefferson Counties in central Oregon. The watershed extends from the crest of the Cascade Mountains to the creek's confluence with the Deschutes River, approximately three miles upstream of Lake Billy Chinook (Figure 2). Elevations range from 10,358 feet at the peak of South Sister to 2,100 feet at the confluence with the Deschutes River.

Snow melt in the Cascade Mountains drives stream flow through Whychus Creek. The high permeability of the surrounding landscape leads to high infiltration and subsurface transport of water (USFS 1998, Gannett *et al* 2001). Associated springs located along the creek, particularly in the Camp Polk and Alder Springs areas, increase flows by 25% to 300%. Tributaries to Whychus Creek include Snow Creek, Pole Creek, and Indian Ford Creek.

Irrigators cumulatively divert up to 90% of the water from Whychus Creek at several points upstream of the City of Sisters. These diversions result in a highly modified stream flow regime that varies greatly depending on the season and the reach. A baseline inventory identified six permanent or seasonal fish passage barriers associated with these diversions blocking upstream fish passage in Whychus Creek from approximately river mile 15 through river mile 27 (UDWC 2009). Fish passage barriers isolate upstream resident fish populations and limit the amount of habitat accessible to anadromous fish.

Land use has impacted fish habitat along Whychus Creek since early European settlers moved into the area. Livestock grazing, urban development, irrigation diversions and other activities have all gradually affected fish habitat quality. In addition, the channelization of 18 miles of creek in the 1960s severely damaged specific reaches (USFS 1998). Channelization, riparian vegetation removal and stream flow modification have reduced the availability of pools, shade, in-stream structure and other important habitat components (USFS 1998).

Restoring anadromous runs to a stream with highly degraded habitat could be a futile effort if stream conditions are unsuitable to support salmonid spawning, rearing, and migration. The 2005 relicensing agreement committed dam operators to investing in passage facilities at the Pelton Round Butte complex, and to habitat restoration upstream of the dams. Fisheries managers introduced the first cohort of more than 200,000 steelhead fry into Whychus Creek in 2007. Additional releases have occurred every year since and will continue according to a jointly developed fish management plan.

Agencies and organizations have embarked on a creek-scale restoration effort in Whychus Creek. Restoration projects slated for the creek range from site-specific land acquisition and channel reconstruction to coordinated barrier removal and stream flow restoration. Restoration practitioners identified a ten-year timeline, beginning in 2009, for implementation of these projects.

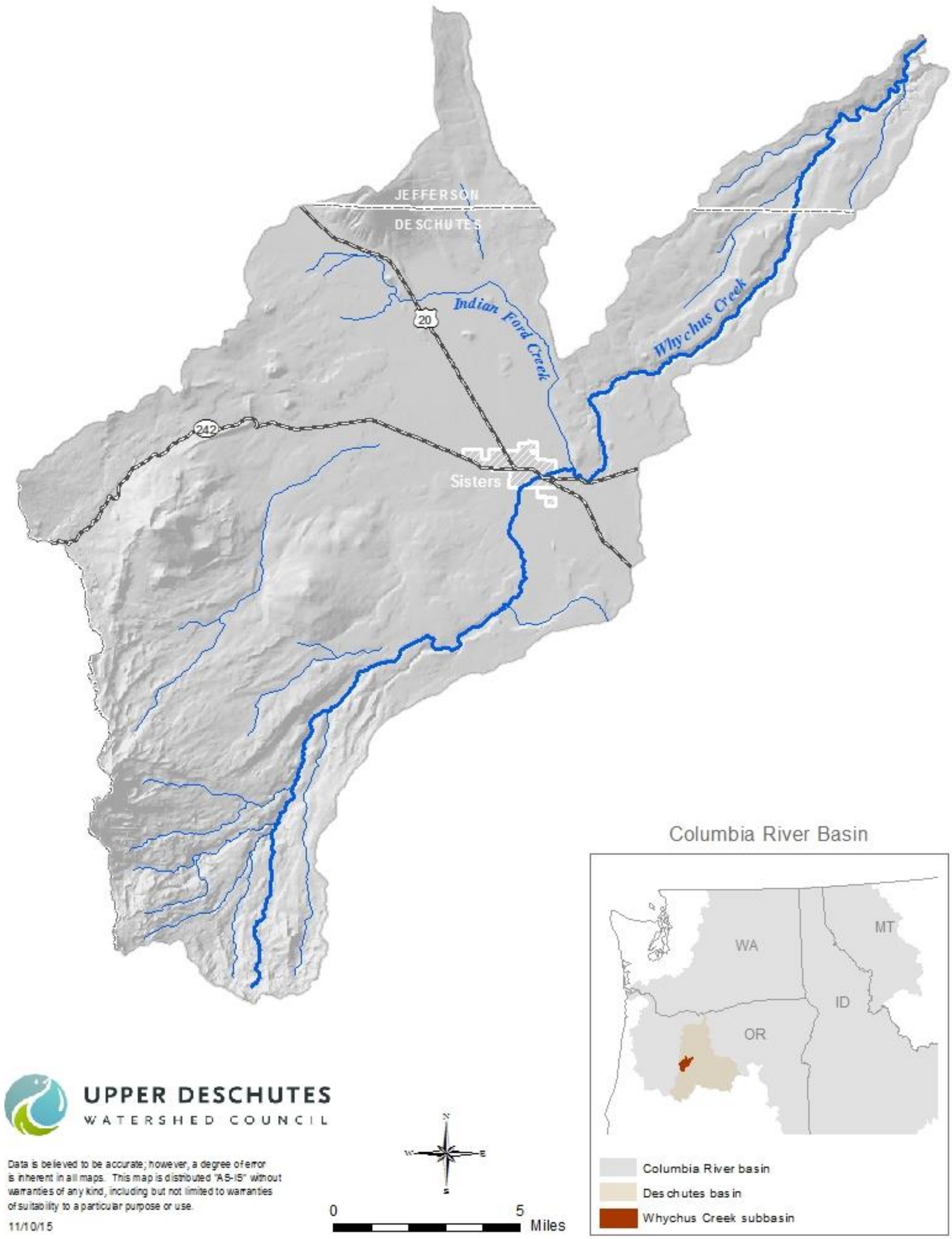

Figure 2.
Whychus Creek extends from the Cascade Range to the Deschutes River. The creek's watershed encompasses approximately 162,000 acres of Deschutes and Jefferson Counties in central Oregon.

Technical Studies

Annual technical studies analyze, interpret, and incorporate available data to examine the status and trends of physical and biological indicators in Whychus Creek. These studies document changes from baseline conditions following the implementation of large-scale habitat restoration actions along the creek, including streamflow restoration, channel restoration and floodplain reconnection, fish passage improvements, screening of irrigation diversions, and other restoration actions. Baseline conditions were inventoried following some streamflow restoration but prior to any other restoration efforts.

Stream flow and stream temperature metrics are anticipated to change in response to stream flow restoration. Golden & Wymore (2016) evaluate low flow metrics to track summer stream flow trends in Whychus Creek from 2000-2014. The 30-day moving average minimum flow represents the lowest flow conditions over the course of a year and indicates when the worst flow conditions occur, contributing to our understanding of when flow restoration is most critically needed. 30-day moving average minimum flows have steadily increased since 2001, most frequently occurring in September (38%; six years), followed by August (31%; five years), and May (19%; three years). The 2015 30-day moving average minimum flow was 14 cfs, from September 16-28.

Mork (2016a) reports on changes in stream temperature in relation to state standards for trout rearing and migration (18°C) and salmon and steelhead spawning (13°C), and discusses the relationship between stream flow and stream temperature in Whychus from 2000 - 2015. Stream temperature (seven day moving average maximum; 7DADM) was warmer than the applicable standard for rearing trout and spawning salmon and steelhead, exceeding the temperature standard at eight sites and for 58% of data days (119 days) at the hottest site on the creek (WC 006.00) in 2015. Stream temperature exceeded the lethal threshold for trout in 2015 for the first time since 2009, for three days at WC 006.00. The 2000 - 2015 stream flow-temperature relationship for WC 006.00 shows 62 cfs of instream flow is needed to meet the 18°C temperature standard in July.

Two reports quantify habitat improvements achieved through passage and screening projects. Mork (2016b) documents the status of stream connectivity along the creek resulting from removal or retrofitting of fish passage barriers as a measure of stream connectivity along the creek. As of 2015, the number of stream miles accessible from the mouth of the creek has increased from the 2009 baseline of 15.5 miles to 26.8; the number of fragmented reaches reduced from seven to two; and the number of barriers reduced from six to one. Mork (2016c) presents reductions in fish entrainment potential on the creek, using unscreened irrigation diversions and the cumulative flows diverted through them as a proxy for entrainment potential. Five diversions were screened or decommissioned between 2009 and 2015; flows diverted through unscreened diversions were reduced by 86%, from 192.9 to 22.0 cfs. Restoration partners plan to provide passage at the last remaining barrier and screen or decommission five of the remaining seven unscreened diversions identified in the 2009 baseline report by 2022. Although stream channel and floodplain restoration projects are expected to improve fish habitat on Whychus, no new habitat data are available to evaluate changes in habitat quality resulting from these projects (but see Mork 2012 for the most current habitat quality ratings for Whychus).

The final two reports update the status of biological conditions in the creek. Mork (2016d) summarizes findings from 2015 PGE O. mykiss and Chinook monitoring. This report outlines the status of O. mykiss populations and Chinook reintroduction. While fish monitoring data from Whychus cannot at present provide information about how restoration is changing stream conditions, they do contribute insight into population dynamics and productivity in Whychus. Density of juvenile O. mykiss has remained consistently low in Whychus (average density in 2015: 30 fish/100m^2) with no significant increasing or decreasing trend despite annual releases of hundreds of thousands of fry; detection of resident O. mykiss redds has fallen substantially since reintroduction began; and 2014 genetic analyses show that the juvenile O. mykiss present in Whychus represent predominantly (75%) Round Butte Hatchery stock, indicating reintroduced fry have residualized in the creek in relatively high numbers. PGE estimated 6,233 naturally-reared steelhead smolts outmigrated from Whychus Creek in 2015, the first year for which a smolt production estimate has been available. Four steelhead (4% of total returns) and seven chinook (14% of returns) ascended the Deschutes River Arm from Lake Billy Chinook in 2015; of these, one of each ascended Whychus Creek.

Mazzacano (2016) examines seven years of macroinvertebrate data to identify trends in macroinvertebrate community composition before and after extensive streamflow restoration. Individual community attributes strongly suggest a community whose composition has changed in response to changing stream conditions, reflecting a greater abundance and/or diversity of taxa that thrive in conditions characterized by cooler temperatures and lower fine sediment, despite other assessments (two indices of biological integrity [IBIs] and a predictive model) that do not indicate substantial improvements in biological conditions in Whychus Creek.

These six reports evaluate improvements in stream conditions in 2015 subsequent to major streamflow, channel and floodplain, and passage and screening restoration projects, as measured by the status of physical and biological indicators. Our intent is that these reports and the data they contain will help restoration partners understand the effectiveness (ecological outcomes) of their actions at moving the creek toward desired conditions. We hope restoration partners continue to draw from these reports to continually improve restoration implementation and monitoring in Whychus Creek.

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Whychus Creek Stream Flow Report

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Abstract

Irrigation diversions in Whychus Creek, a tributary to Oregon's Deschutes River, historically diverted up to 100% of the flow from the creek during the summer irrigation season. Restoration partners have focused on restoring summer stream flow in the creek to support the reintroduction of steelhead trout and Chinook salmon. The Deschutes River Conservancy (DRC) used stream gage data from Whychus Creek to determine the status of selected stream flow metrics prior to and during large scale stream restoration along the creek. Three metrics characterize low flows in the creek. The minimum 30 day moving average flow represents annual low flow conditions. May median flow represents late spring/early summer conditions. August median flow represents late summer conditions. Minimum 30 day moving average flows generally occurred in August and early September from 2000 through 2010. From 2011 through 2015 minimum 30 day moving average flows occurred in late May (2011 and 2012), late July (2013), October (2014), and September (2015). Annual minimum 30 day moving average flows increased or remained constant in every year except for 2005, 2009, and 2015. May median flows exhibited both inter-annual and intra-annual variation. May median flow ranged from a low of 5.4 cfs in 2003 to a high of 64 cfs in 2012. August median flows also exhibited inter-annual and intra-annual variation but intra-annual variation was typically lower than in May. August median flow ranged from a low of 2.6 cfs in 2002 to a high of 35 cfs in 2014. These results suggest that Whychus Creek still experiences low flows during both late spring/early summer and late summer/early fall flow, two periods when irrigation demands generally exceed water availability. Extreme flows, however, appear to be decreasing in magnitude during both of these periods. The relationship between stream flow and stream temperature from 2000-2015 clearly shows that low flows during late summer/early fall result in very poor temperature conditions for native trout; low late spring/early summer flows result in temperatures too warm for steelhead spawning, and may compromise other important habitat functions. These results highlight the need to redouble efforts to restore significantly more stream flow in Whychus Creek during late summer/early fall, and to better understand how low flows during late spring/early summer may limit ecosystem function. Securing state instream water rights that adequately represent the minimum stream flow necessary to support reintroduced anadromous and native resident salmonid populations is one approach to facilitate stream flow restoration beyond the 33 cfs for which state instream water rights currently exist in Whychus Creek. As restoration continues to increase flows in Whychus Creek, restoration partners should continue to evaluate both early and late season flow as well as extreme low flows to fully describe restoration outcomes.

Introduction

Stream habitat alteration occurs in two different ways: human disturbances directly alter stream habitat and human disturbances also prevent natural disturbances from occurring. Both types of disturbance alter stream habitat (NRC 2002). Irrigation diversions along Whychus Creek diverted up to 90% of the creek's flow from April through October during the study period (Figure 1) and cause both of these types of disturbances. Restoration partners have identified these stream flow alterations as a primary factor limiting fish production in Whychus Creek.

The entire hydrograph affects what a stream looks like and how it functions (Poff *et al* 1997). Different components of the hydrograph may drive different ecological processes (Doyle *et al* 2005). Changes in stream flow can affect biological characteristics such as macroinvertebrate assemblages (Dewson *et al* 2008, Konrad *et al* 2008, James *et al* 2008, Monk *et al* 2008, Wills *et al* 2006), fish communities (Xenopoulos *et al* 2006, Decker e*t al* 2008), and riparian vegetation (Stromberg *et al* 2005). By removing up to 90% of the stream flow from Whychus Creek, irrigation diversions have eliminated all but the low flow components of the hydrograph during the summer and likely affected each of these characteristics. Monitoring the status and trends of stream flow in Whychus Creek will illuminate whether the stream is moving towards or away from desired conditions.

Hydrologists have developed a wide range of hydrograph-related metrics to track stream flow conditions over time. These different metrics relate to different components of the hydrograph that affect physical and biological conditions in a stream. Olden and Poff (2003) identify 171 metrics that appeared in 13 papers. These metrics relate to the magnitude, frequency, rate of change, duration, or timing of flow events. Monk *et al* (2007) built off of Olden and Poff (2003) to identify an additional 30 metrics. Others have attempted to identify a subset of metrics that represent hydrologic alteration across a wide range of conditions (Olden and Poff 2003, Monk *et al* 2007, Yang *et al* 2008, Gao *et al* 2009). Researchers have not yet identified a single subset of metrics that represent alteration in all types of streams. Different types of streams have different hydrologic characteristics. For example, groundwater-dominated streams exhibit relatively low seasonal variability while snowmelt-dominated streams exhibit clear seasonal patterns. The type of stream, surrounding geography, and the desired conditions in that stream define the appropriate set of metrics.

This study focuses on low flow metrics that relate to expected stream flow restoration. Pyrce (2004) identifies and categorizes low flow indices from published and unpublished sources. Many of these focus on seven day averages and their exceedances. Although these metrics appear to be widely used across the United States, they were originally intended for specific purposes such as water quality regulation and may not be appropriate for the identification of ecological flows (Pyrce 2004).

This study uses three metrics selected from the Indicators of Hydrologic Alteration that represent flow magnitude and timing (Richter *et al* 1996, Table 1). Generally, flow magnitude relates to habitat availability within a stream or river (Richter *et al* 1996). However, flow timing also affects habitat availability. Yang *et al* (2008) studied the relationship between fish communities and flow in the Illinois River. Their results suggest that low flow timing affects fish diversity while low flow magnitude affects overall abundance.

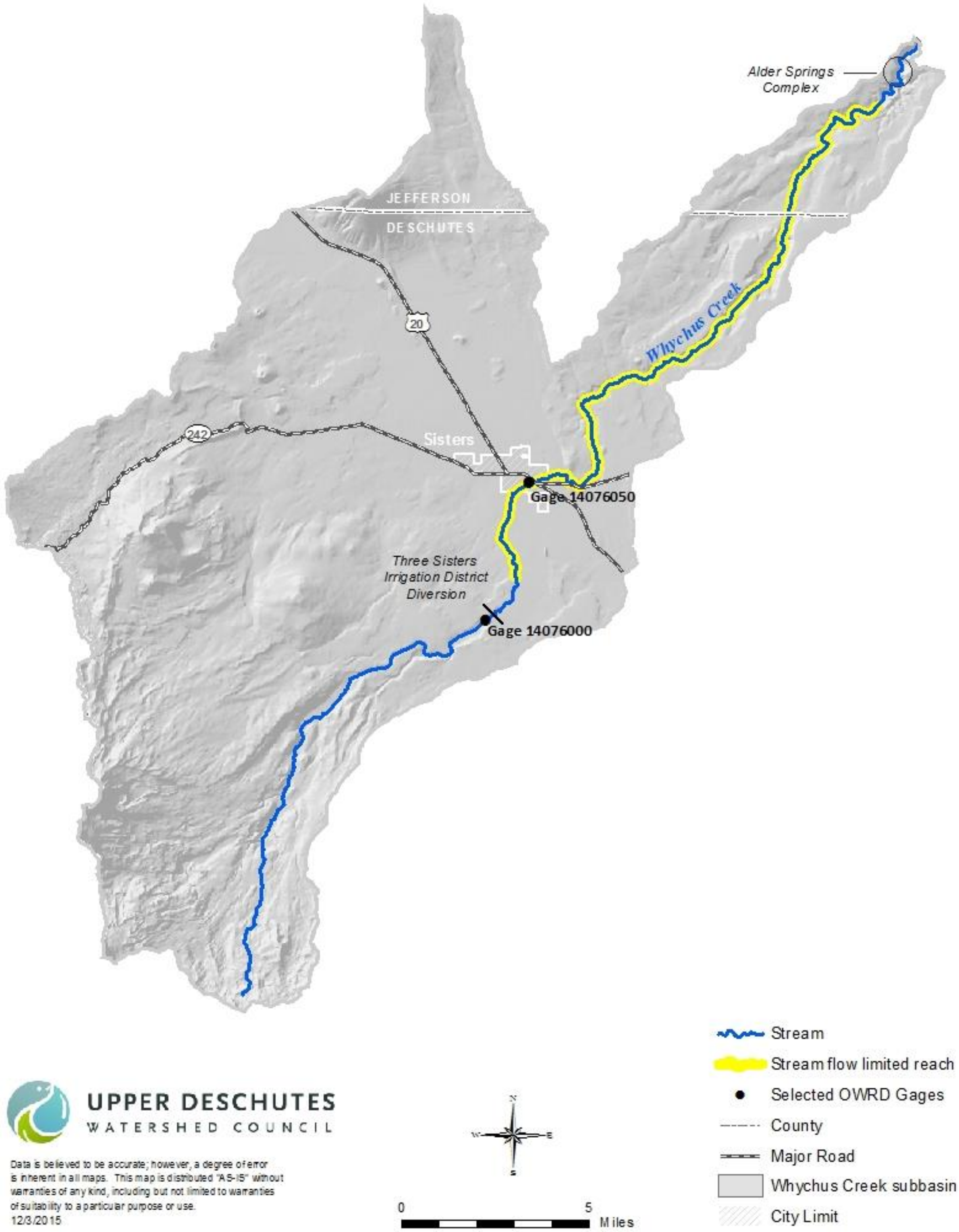

Figure 1.
Stream flow limits stream function in Whychus Creek downstream from the Three Sisters Irrigation District Diversion. Spring inputs near the mouth of Whychus Creek increase stream flow and improve conditions in the creek.

Metric	Appears In
30 day minimum	Gao *et al* 2009, Richter et al 1996
May median flow	Gao *et al* 2009, Richter et al 1996
August median flow	Richter *et al* 1996

Table 1.
The three metrics selected for this report relate to the stream flow restoration goals identified by restoration partners. They represent discharge magnitude and timing during low flow periods.

The status and trends of these metrics will inform restoration partners about the effectiveness of stream flow restoration. These metrics do not represent the entire hydrograph. Instead, they represent conditions in the creek during the summer irrigation season. Irrigation diversions alter flows more during this period than during other times of year. Restoration partners have addressed and expect to address primarily low summer flows over the next ten years. The existing legal framework surrounding stream flow restoration, combined with a lack of storage reservoirs along the creek, hinders the restoration of other components of the hydrograph.

Minimum 30 Day

The minimum 30 day moving average flow generally represents annual low flow conditions in Whychus Creek. As Richter *et al* (1996) note, life stages of aquatic organisms often link to hydrologic cycles. Changes in the timing and magnitude of the minimum 30 day moving average flow may affect these organisms. Restoration partners expect both the timing and magnitude of this metric to change as a result of restoration activities in Whychus Creek.

May Median

May median flow may provide a general indicator of spawning habitat availability in Whychus Creek. Redband trout spawning in the Deschutes Basin centers on the month of May (Oregon Department of Fish and Wildlife 2005). Increasing irrigation demands prior to peak runoff typically stress water supplies in the creek during this period. Restoration partners expect to increase May stream flows through water transactions with irrigators.

Richter *et al* (1996) suggest the use of mean monthly flows to characterize the central tendency of stream flows. Median monthly flows provide a similar measure of central tendency that minimizes the influence of outliers (Helsel and Hirsch 2002). Using the median instead of the mean may provide a better measure of central tendency when human actions lead to outliers such as extreme low or high flow events.

August Median

August median daily average flow provides an indicator of late summer flow availability in Whychus Creek. Decreasing snow pack and steady irrigation demands typically stress water supplies in the creek during this period and stream flow often reaches a nadir. Low flow magnitude provides one measure of habitat availability during this period (Richter *et al* 1996).

Methods

Data Collection

The Oregon Water Resources Department (OWRD) maintains several gages along Whychus Creek. They operate gage 14076050 at the City of Sisters, downstream from major irrigation diversions along the creek (Figure 1). OWRD began operating this gage in 2000 and has continued operating it through the publication of this report in 2016. This report uses data from this gage. OWRD operates another gage, 14075000, upstream from all diversions on Whychus Creek. They have published stream flow data for this gage from 1906 through 2016. Why not estimate historic stream flows at the City of Sisters over a longer time period for these analyses? Water transactions for stream flow restoration in Whychus Creek occurred during every year of the study period. Conditions through the study period are neither static nor represented by historic conditions. The period from 2000 through 2015 reflects conditions in the creek during ongoing restoration efforts.

Gage 14076050 records stream stage in Whychus Creek at Sisters, OR. The gage consists of a float-tape system that records stream stage every fifteen minutes (Burright A. Personal communication. August 24, 2009). OWRD obtained preliminary data from this gage on a near-real time basis through an automated, remote telemetry-based process. OWRD reviewed this data based on their knowledge of site conditions and site-specific stage-discharge relationships. They estimated any missing values and revised any values believed to be erroneous (OWRD 2009). OWRD reviewed this data again before publishing it as daily average discharge data online. OWRD had published final data from May 18, 2000 through September 30, 2008 and from October 1, 2009 through September 30, 2011 when this report was prepared. OWRD had released provisional data from October 1, 2008 through September 30, 2009, October 1, 2011 through October 31, 2012, and from November 1, 2013 through September 30, 2015 when this report was prepared.

Data Analysis

The Deschutes River Conservancy (DRC) analyzed published or provisional stream flow data for gage 14076050. The DRC analyzed this data for each water year, extending from October 1 through September 30, between 2000 and 2015. OWRD installed this gage in 2000 and only published data for the 2000 water year after May 17. All analyses except for the August median flow omitted year 2000 due to incomplete data.

<u>Minimum 30 Day</u>

The DRC used spreadsheet software to determine the timing and magnitude of the minimum 30 day moving average flow at gage 14076050. The DRC considered each water year independently. Moving averages extended to 14 days before and 15 days after the date for which the value was being calculated. Initial data exploration suggested that low flow periods extended across water years. Dividing the data by water year, October 1 through September 30, did not fully represent the low flow periods experienced each season. The DRC used an extended water year, November 1 through October 31, to capture low flow periods that extended across water years. The DRC completed this analysis for extended water years 2001 through 2015.

<u>May Median</u>

The DRC used spreadsheet software to determine the median daily average flow during the month of May for years 2001 through 2015. The DRC only had partial data for 2000 and did not include that data in this analysis.

August Median

The DRC used spreadsheet software to determine the median daily average flow during the month of August for years 2000 through 2015. The DRC had complete data for August 2000 and included that data in this analysis.

Results

Minimum 30 Day

The minimum 30 day moving average discharge at the Oregon Water Resources Department's gage number 14076050 generally occurred during August and early September (Table 2). This discharge ranged from 2.40 cfs in 2002 to 22.00 cfs in 2013 and 2014. It increased or remained constant each year except for 2005, 2009, and 2015.

Year	30 Day Minimum (cfs)	Dates
2001	2.55	9/25/2001 – 9/27/2001
2002	2.40	8/8/2002 – 8/14/2002
2003	3.60	9/19/2003 – 10/1/2003
2004	8.15	8/6/2004 – 8/18/2004
2005	6.70	8/4/2005 – 8/11/2005, 8/15/2005 – 8/19/2005
2006	12.00	8/24/2006 – 8/27/2006
2007	12.00	8/28/2007 – 8/31/2007
2008	16.00	4/25/2008 – 5/7/2008, 9/7/2008 – 9/30/2008
2009	13.00	9/14/2009-9/22/2009
2010	19.00	9/1/2010-9/13/2010
2011	21.00	5/19/2011 – 5/23/2011
2012	21.00	5/24/2012
2013	22.00	7/19/2013-7/31/2013
2014	22.00	10/6/2014-10/7/2014
2015	14.00	9/16/2015-9/28/2015

Table 2.
The minimum 30 day moving average discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 provides one indicator of low flow magnitude and timing.

The 2015 minimum 30-day moving average ranked 9th of fifteen years of minimum 30-day flow values, behind every year since 2008 except for 2009 (Figure 2). The 2015 minimum 30-day discharge was slightly above half the minimum flow value in 2013 and 2014.

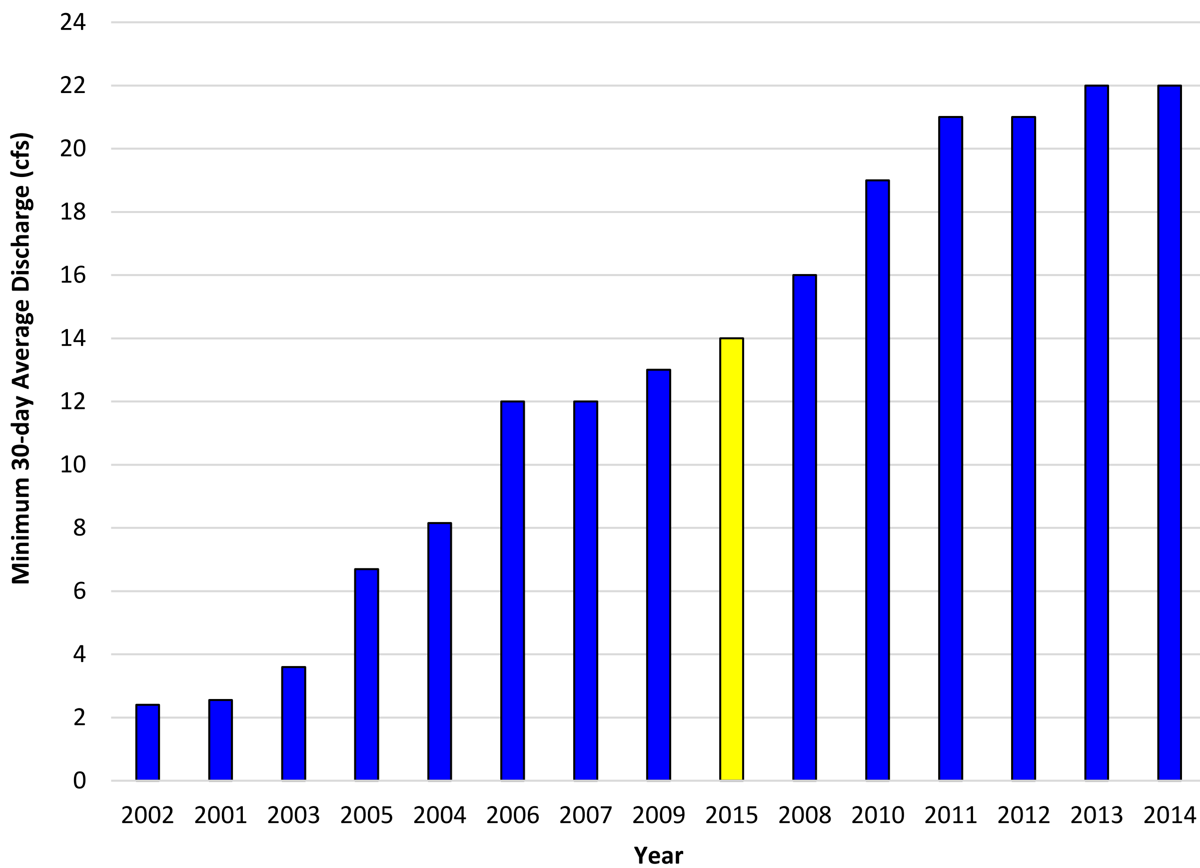

Figure 2. The 2015 minimum 30-day moving average discharge ranked 9th of fifteen years of minimum 30-day flow values, behind every year since 2008 except for 2009.

May Median

The DRC analyzed stream flow data as described above. Average May flow in Whychus Creek at the Oregon Water Resources Department's gage number 14076050 exhibited both inter-annual and intra-annual variation (Figure 3). Median flow during the month of May ranged from a low of 5.4 cfs in 2003 to a high of 64.0 cfs in 2012. 2006 exhibited the greatest intra-annual variation in May flow, with a 20th percentile value of 22.0 cfs and an 80th percentile value of 122.0 cfs.

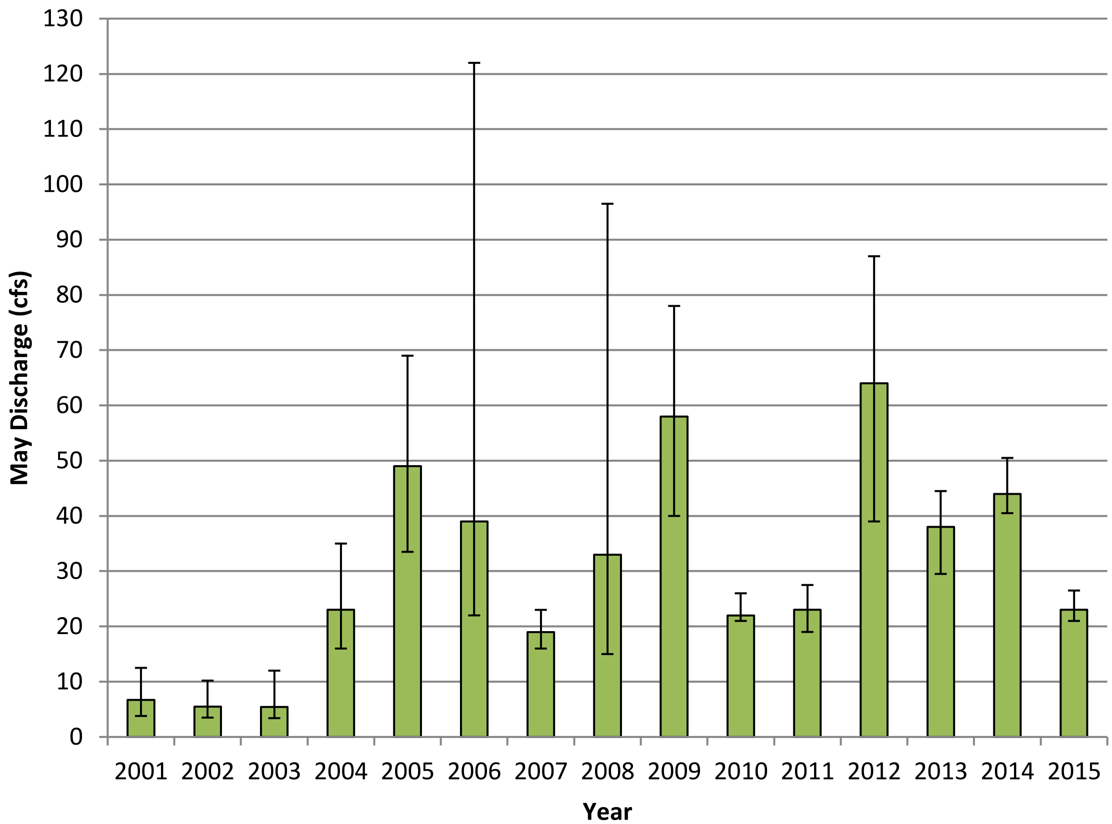

Figure 3.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of May provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of May at this location.

August Median

Median discharge during the month of August exhibited both inter-annual and intra-annual variation at gage number 14076050 (Figure 4). 2002 exhibited the lowest median discharge during the month of August, with a median daily average discharge of 2.6 cfs. 2011 exhibited the highest median discharge during the month of August, with a median daily average discharge of 32.0 cfs. 2011 also exhibited the greatest intra-annual variation in discharge, with a 20th percentile discharge of 27.5 cfs and an 80th percentile discharge of 45.0 cfs.

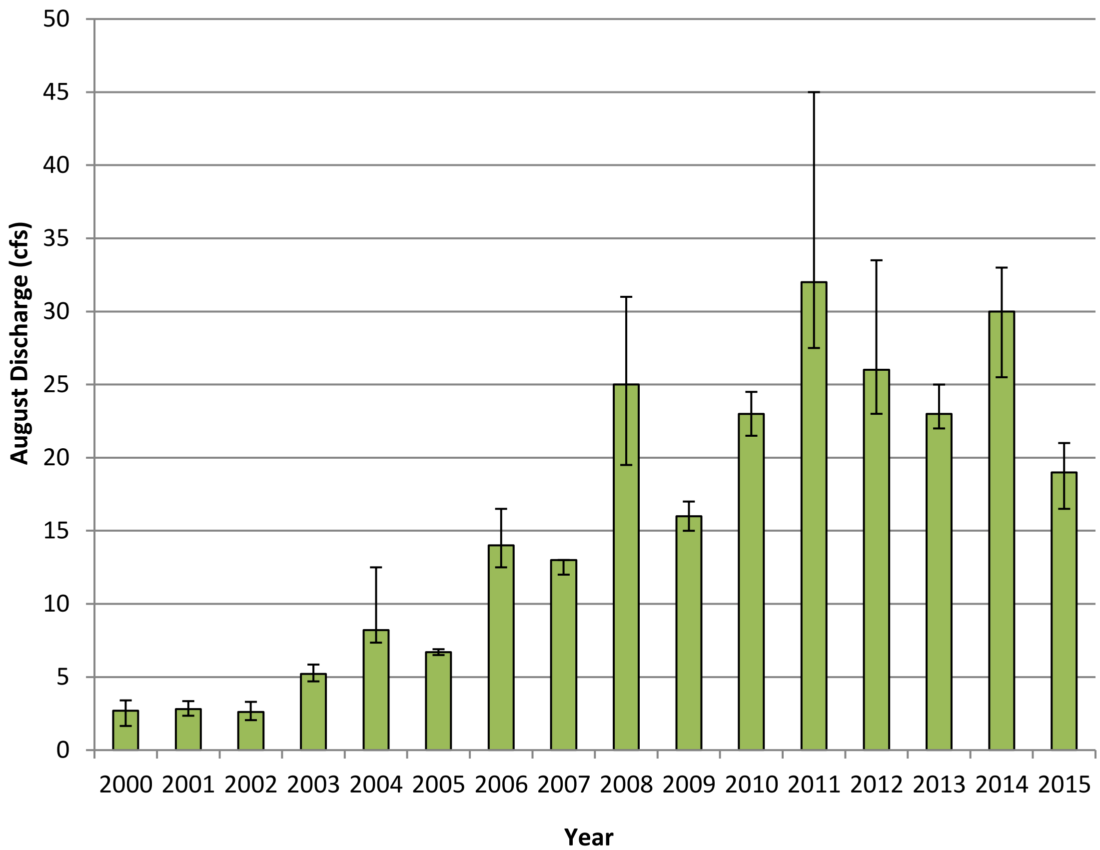

Figure 4.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of August provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of August at this location.

Discussion

The analyses in this report describe stream flow conditions in Whychus Creek across fifteen years of stream flow restoration. They focus on the period from 2000 through 2015. Restoration partners have prioritized the restoration of summer base flow in Whychus Creek downstream from the Three Sisters Irrigation District diversion. The three metrics included in this report characterize low flow conditions in Whychus Creek. These metrics suggest that flow lows continue to occur in both late spring/early summer and late summer/early fall but that the magnitudes of these low flows are greater (i.e. flows are higher) than in the past. Minimum 30-day moving average data show flow lows occurring during various months throughout the irrigation season, with 2008-2015 lows occurring during April, May, July, September, and October.

Minimum 30 Day

The minimum 30-day moving average increased in most years since 2001 but decreased in 2004, 2009, and, most significantly, 2015. Whereas in 2013 and 2014 the minimum 30-day moving average discharge, representing *minimum annual low flow conditions* for both years, was 22 cfs, in 2015 average daily flow exceeded 20 cfs for only five days out of two months between mid-August and mid-October. The exceptionally low flows observed in 2015 very likely reflect the approximately 10% increase in infiltration that occurred in the Whychus Floodplain restoration project between the TSID irrigation diversion and the City of Sisters OWRD stream flow gage in 2015 (Jonathan LaMarche, personal communication, October 2016), rather than an increase in flow diverted for irrigation at TSID. Recent years have seen the timing of annual low flows occurring more often in the spring (May), although 2014 and 2015 lows both occurred in the fall.

Late Spring/Early Summer Flows

May daily average stream flow results continue to display a wide range of inter-annual and intra-annual variability (Figure 2). Although August monthly median flows tend to be lower than May monthly median flows (Figure 2, Figure 3), May monthly median flows appear to exhibit greater intra-annual variability.

Instream water rights awarded to the State of Oregon in the 1990s to support fish populations provide an interim base flow target. Median daily average flow during the month of May exceeded Oregon's 20 cfs instream water right for Whychus Creek upstream from Indian Ford Creek in eleven out of fifteen years (OWRD 1996, Figure 2), and met Oregon's March, April, and May instream water right of 50 cfs for Whychus Creek downstream from Indian Ford Creek in two of fifteen years (OWRD 1996, Figure 3).

Restoration partners have focused on late summer stream flow as a metric for restoration effectiveness. Late spring/early summer stream flow is very likely as important for stream function. As noted earlier, redband trout spawning centers on the month of May (ODFW 2005); steelhead spawning and emergence is anticipated to occur from April through June. Low flows during late April, May, and early June result in stream temperatures that regularly exceed the 13°C required for steelhead spawning and emergence, and sometimes exceed the 18°C redband trout require year-round for rearing. Beyond the effects of stream temperature, low flows during these months may limit available spawning habitat (quantity), and extreme low flow events during this period may limit fish production by dewatering existing redds. Results suggests that extreme low flows occur less often during this period than they have in the past, consistent with efforts to restore base flows to Whychus Creek.

Late Summer/Early Fall Flows

This analysis suggests that, although Whychus Creek continues to experience low flows during late summer and early fall, flows during this period have improved. The annual minimum 30 day moving average stream flow occurred during the month of August or September in eleven of fifteen years included in this study through 2015 (Table 2). Although stream flow naturally decreases during this period, even in the worst water years August flows upstream of diversions have consistently been above 70 cfs, sufficient to protect suitable stream temperature and, presumably, other habitat functions of stream flow beyond temperature. Given the amount of flow in August upstream of diversions, the magnitude and frequency of late summer/early fall flows in Whychus Creek below irrigation diversions suggest low flows likely limit fish populations.

The State of Oregon instream water right again provides an interim base flow target in Whychus Creek. Median daily average flow during the month of August exceeded Oregon's 20 cfs instream water right for Whychus Creek upstream from Indian Ford Creek in 2008 and 2010-2014 (OWRD 1996a, Figure 3). It never met the state instream water right of 33 cfs for Whychus Creek downstream from Indian Ford Creek (OWRD 1996b, Figure 3). Late summer and early fall base flows continue to fall short of these targets, resulting in stream temperatures that are too warm for native salmonids and very likely limit these populations; low late summer and early fall flows likely also limit habitat functions beyond stream temperature. Increasing these flows should remain a high priority for restoration partners, who should continue to evaluate when late summer/early fall low flows occur and use August or September median flows as an indicator of restoration effectiveness.

Recommended Actions

Restoration partners have focused on restoring base flows to this historically dewatered stream system. They have operated under the assumption that base flows are critical to providing the habitat necessary to support self-sustaining populations of anadromous fish. They used, and continue to use, the instream water rights awarded to the State of Oregon as interim stream flow targets. Legally protected stream flows are currently approaching the June through February state instream water right of 33 cfs in Whychus Creek. The reliability of these water rights varies based on water availability in Whychus Creek, leading to inter- and intra-annual variability in the low flow metrics discussed earlier. Restoration partners should continue to evaluate these low flow metrics to understand how stream flow in Whychus Creek is changing with restoration actions. Evaluating additional extreme low flow metrics may further inform restoration partners as to the success of their actions.

Stream flow and stream temperature data from Whychus Creek (Mork 2016) clearly show the state instream water right of 33 cfs downstream of Indian Ford Creek is insufficient in June through September to provide suitable habitat conditions for native salmonids. These data point to an evident need for state instream water rights to adequately represent the minimum stream flow necessary to support reintroduced anadromous and native resident salmonid populations. Securing these water rights will require a qualified entity applying for them.

Restoration partners have not focused on restoring non base-flow components of this hydrograph beyond base flow. High stream flow events continue to occur before, during and after the irrigation season. Irrigation operations have reduced the magnitude of but not eliminated these events. Although describing a desired hydrograph would better inform restoration partners about the status of the stream flows in Whychus Creek, it would likely not improve the ability of restoration partners to address other hydrograph components. The absence of any storage reservoirs along Whychus Creek and the continued presence of high flow events have reduced the priority of evaluating non-base flow components of the hydrograph.

The three stream gages operated by the Oregon Water Resources Department on Whychus Creek measure flow above all irrigation diversions, below most irrigation diversions, and below natural spring inputs at 15 minute intervals. Currently, OWRD only publishes daily average stream flow at each of their gages. Daily average flows do not fully represent the range of flows in Whychus Creek; they mask diurnal fluctuations and may not reveal low or high flow peaks. Past reports recommended the use of 15-minute flow data in outcome evaluation. Fifteen-minute interval data may more precisely describe conditions in the creek but it is not necessarily accurate as it does not go through OWRD review and publication. Due to potential inaccuracies in this data, restoration partners expect to continue using this 15-minute interval data for real time evaluation of stream flows but not for long-term outcome evaluation.

Acknowledgements

The Oregon Water Resources Department provided the data necessary to complete this report. Their extensive gaging network and published data were critical to its production. The Bella Vista Foundation, Bonneville Environmental Foundation, Laird Norton Family Foundation, National Fish and Wildlife's Columbia Basin Water Transactions Program, and Oregon Watershed Enhancement Board have

supported the monitoring and evaluation necessary to understand restoration effectiveness in Whychus Creek.

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Whychus Creek Water Quality Status, Temperature Trends, and Stream Flow Restoration Targets

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Abstract

Diversion of almost 90% of summer stream flow and channelization of over 50% of the length of Whychus Creek have degraded water quality, resulting in Whychus Creek running dry two out of three years from 1960 until 1998, and an ODEQ listing of water quality limited since 1998. The Upper Deschutes Watershed Council monitored temperature from 2000 through 2015 at eleven sites representing diverse flow conditions in Whychus Creek. This report incorporates 2015 data to 1) evaluate the 2015 status of stream temperature in Whychus Creek relative to state standards for salmonid spawning, rearing and migration; 2) quantify 2000-2015 temperature trends in relation to stream flow; 3) describe the effects of stream flow and air temperature on stream temperature in Whychus Creek, and 4) update temperatures predicted to occur at the observed range of Whychus Creek stream flows. Temperatures exceeded the state rearing and migration temperature standard of 18˚C at eight monitoring sites and exceeded the January 1 – May 15 13˚C steelhead spawning criteria at nine monitoring sites in 2015; temperatures exceeded the 13˚C spawning criteria during the anticipated September Chinook salmon spawning period at eight sites in 2015. Stream temperature exceeded rearing and migration and steelhead spawning criteria for 58% of data days (119 days) at WC 006.00 (FS Road 6360) in 2015, a higher percentage of days than in the previous six years and higher than in all but five of the thirteen total years for which data are available, and exceeded 13˚C in September for 100% of data days (30 days). Stream temperature exceeded the 24˚C lethal threshold for trout in 2015 for the first time since 2009, for three days at WC 006.00. Regression of 2000-2015 temperature and flow data and Heat Source model results show more than 60 cfs is required to meet 18°C on average in lower reaches of Whychus Creek in July; stream temperatures as high as 21°C are predicted to occur at 63 cfs, emphasizing the imperative need for 60 cfs as a minimum flow during July to reduce stream temperatures below the threshold at which trout experience chronic effects that result in mortality. These results show the 33 cfs state water right to be far short of the flows needed to meet the state temperature standard or provide suitable conditions for fish in downstream reaches of Whychus Creek throughout the irrigation season. Continued development of creative solutions to allocate flow instream in Whychus Creek in low water years is needed to guarantee conditions that will support the recovery of reintroduced native fish populations. These results contribute to an improved understanding of temperature and flow that we hope will support restoration partners in planning more ambitious stream flow restoration efforts on Whychus Creek.

Introduction

Restoration partners have identified the Whychus Creek watershed as a priority watershed for conservation and restoration within the upper Deschutes Basin (NWPPC 2004, UDWC 2006). Diversion of almost 90% of average summer flows and historic channelization of nearly 50% of the creek length have created conditions that contribute to elevated stream temperatures and may compromise other water quality parameters. Whychus Creek has been listed by ODEQ under Clean Water Act Section 303(d) as water quality limited with TMDLs needed for temperature (Category 5) and categorized as having insufficient data for assessment for dissolved oxygen and pH (Category 3; Table 1, Figure 1) since 1998.

UDWC began monitoring temperature on Whychus Creek in 1995. In 1999 DRC stream flow restoration efforts first returned continuous summer flows to Whychus Creek, and the volume of flows protected instream has incrementally increased since. Restoration partners expect that increasing stream flow will reduce temperatures in Whychus Creek to more frequently and consistently meet spawning and rearing and migration habitat requirements for native fish including anadromous steelhead trout and Chinook salmon re-introduced to the creek in 2007 and 2009, respectively.

Water temperature affects the growth and survival of aquatic organisms. Temperature naturally fluctuates on a daily and seasonal basis, with daily fluctuations resulting from continuous changes in solar radiation and air temperature, and seasonal fluctuations in response to changes in climate, solar aspect, and variable amounts of stream flow from snowmelt and precipitation. Water temperature naturally increases as water flows downstream, and temperature can decrease as a result of groundwater inflows (springs) or the inflow of cooler tributaries. Anthropogenic changes that alter the natural hydrograph, such as diversions for irrigation, groundwater pumping, and climate change, also influence temperature.

ODEQ state temperature standards were developed to protect fish and other aquatic life in Oregon waterways (ODEQ 2009). The year-round temperature standard applied to Whychus Creek for salmon and trout rearing and migration specifies that seven-day moving average maximum (7DADM) temperatures are not to exceed 18˚C. The 2002 303(d) list also identified Whychus Creek as not meeting the 13˚C state temperature standard for salmon and steelhead spawning. No subsequent 303(d) list has applied this criterion to Whychus Creek because anadromous fish were not spawning in Whychus Creek when data for these lists were collected. However, this habitat use is anticipated to resume, and the spawning temperature standard to become relevant, as steelhead and salmon reintroduced in 2007 and 2009 begin to return to the creek. The State of Oregon 1992-1994 Water Quality Standards Review (ODEQ 1995) identified 24˚C as the lethal temperature threshold for salmon and trout. Runge et al (2008) showed stream temperatures as low as 20˚C to have chronic sub-lethal effects on rainbow trout, with trout survival inversely related to the amount of time stream temperatures were 20˚C. Twenty-two degrees Celsius (22˚C) is generally agreed to have severe consequences for trout, including decreased foraging and increased aggressive behavior (Nielsen 1994), elimination of salmonids from a location (Nielsen 1994, US EPA 1999), and broad mortality (US EPA 2003). For steelhead and Chinook salmon spawning conditions, egg mortality is high at 15˚C compared to lower temperatures (Myrick and Cech 2001).

In addition to temperature, dissolved oxygen and pH levels also directly affect aquatic organisms. Waterways naturally produce oxygen through photosynthesis and aeration. Dissolved oxygen is consumed through respiration and degradation of organic plant compounds. The amount of dissolved oxygen available (percent saturation) is also affected by altitude and temperature: water at higher altitudes holds less dissolved oxygen than water at lower altitudes (because the degree of atmospheric pressure is less at higher altitudes), and cold water holds more dissolved oxygen than warm water. When oxygen is consumed at a faster rate than it is produced, dissolved oxygen concentrations fall, negatively affecting aquatic organisms. Salmon and trout, especially in their early life stages, are very susceptible to low dissolved oxygen concentrations.

Water pH levels (alkalinity) are primarily affected by plant photosynthesis, but can also be influenced by the chemistry of the local substrate. The volcanic soils of the Upper Deschutes Basin may increase the acidity (and decrease pH) of basin waterways. Water pH directly influences aquatic insect populations as well as salmon and trout egg development, egg hatching, and embryo development. Extreme pH levels can negatively impact fish by increasing the availability and toxicity of pollutants such as heavy metals and ammonia.

Whychus Creek is categorized as having insufficient data for assessment for dissolved oxygen and pH. UDWC analyses of dissolved oxygen data collected from 2006 to 2008 indicated that Whychus Creek met state dissolved oxygen standards for salmon and trout rearing and migration, although dissolved oxygen levels did not consistently meet state criteria for salmon and trout spawning (Jones 2010). Because dissolved oxygen saturation is directly affected by temperature, we expect dissolved oxygen levels to track temperature trends. While observed trends in stream temperature continue to demonstrate cooling, and in the absence of other novel environmental conditions, we expect dissolved oxygen levels to improve or remain constant. Under these circumstances, temperature data are a suitable proxy for dissolved oxygen data, and indicate dissolved oxygen levels that will continue to meet the state standard for salmon and trout rearing and migration. UDWC discontinued monitoring dissolved oxygen on Whychus Creek in 2009 on this premise. A consistent warming trend in temperature would flag potentially deteriorating dissolved oxygen conditions and warrant resuming monitoring of dissolved oxygen. Although 2006-2008 data indicated pH standards were not consistently met in the summer, low pH values were attributed to the influence of volcanic soils and were not expected either to limit ecological function or to be affected by increased flows with stream flow restoration. Accordingly we also discontinued monitoring pH subsequent to 2009. While this report does not present dissolved oxygen or pH data, we consider the observed trends in temperature to provide a surrogate measure of water quality in Whychus Creek. For further discussion of temperature, dissolved oxygen, pH, and state standards for each parameter, refer to *Whychus Creek Water Quality Status, Temperature Trends, and Stream flow Restoration Targets* (Jones 2010).

The stream flow and habitat restoration efforts of Deschutes River Conservancy (DRC), UDWC, and restoration partners aim to improve water temperatures to meet the18˚C state standard and support sustainable anadromous and resident native fish populations by reducing warming rates and reconnecting the creek to floodplains and groundwater. DRC and restoration partners identified a stream flow target for Whychus Creek according to state instream water rights. State of Oregon March, April and May instream water rights protect 20 cfs upstream and 50 cfs downstream of Indian Ford Creek (RM 18); state water rights for June and July, and for August and September when flows are historically low, specify 20 cfs upstream and 33 cfs downstream of Indian Ford Creek. Protected water rights correspond to recommended minimum flows identified through the Oregon Method, which relates stream flow to fish habitat availability (Thompson 1972), however UDWC analyses and the HeatSource model (Watershed Sciences and MaxDepth Aquatics 2008) have shown these flows to be insufficient to create suitable conditions for fish or meet state temperature standards. The DRC stream flow restoration target aims to protect 33 cfs instream at Sisters City Park. Because no substantial flows enter Whychus Creek between this location and Alder Springs just below WC 001.50, the DRC target will effectively also protect 33 cfs downstream of Indian Ford Creek.

While previous regression analyses of stream temperature and stream flow in Whychus Creek have shown that stream flow explains a high proportion of the variation in stream temperature, wide prediction intervals for fitted stream temperatures suggest another environmental variable influencing stream temperature. To better understand the variability in stream temperature that is not explained by stream flow and explore how air temperature affects observed stream temperatures in Whychus Creek, we included air temperature in 2015 regression analyses.

This report presents analyses of 2000-2015 temperature and flow data that: 1) evaluate the 2015 status of stream temperature in Whychus Creek relative to state standards and anticipated timing for salmonid spawning, rearing and migration; 2) quantify temperature trends in relation to stream flow; 3) describe the effects of stream flow and air temperature on stream temperature in Whychus Creek, and 4) update temperatures predicted to occur at the observed range of Whychus Creek stream flows.

Table 1. 2012 Oregon Clean Water Act Section 303(d) status of Whychus Creek.

	Parameter	**Temperature**		**Dissolved Oxygen**		**pH**	
	Beneficial Use	Salmon & Trout Rearing & Migration	Steelhead Spawning	Salmon & Steelhead Non-Spawning	Salmon & Trout Spawning	Multiple Uses	Multiple Uses
	Season	Year Round	January 1 - May 15	Year Round	January 1 - May 15	Fall/ Winter/ Spring	Summer
	Standard	18° C	13° C	8.0 mg / L @ 90% Sat	11.0 mg / L @ 90% Sat	6.5-8.5 SU	6.5-8.5 SU
ODEQ Reach	0 - 40.3	TMDL Needed	Not Applicable	Not Applicable	Insufficient Data for Section 303(d) Assessment	Insufficient Data for Section 303(d) Assessment	Insufficient Data for Section 303(d) Assessment
	1 - 13.3	Not Applicable	Not Applicable	Insufficient Data for Section 303(d) Assessment	Not Applicable	Not Applicable	Not Applicable
	13.3 - 40.3	Not Applicable	Not Applicable	Insufficient Data for Section 303(d) Assessment	Not Applicable	Not Applicable	Not Applicable

Source: ODEQ 2016

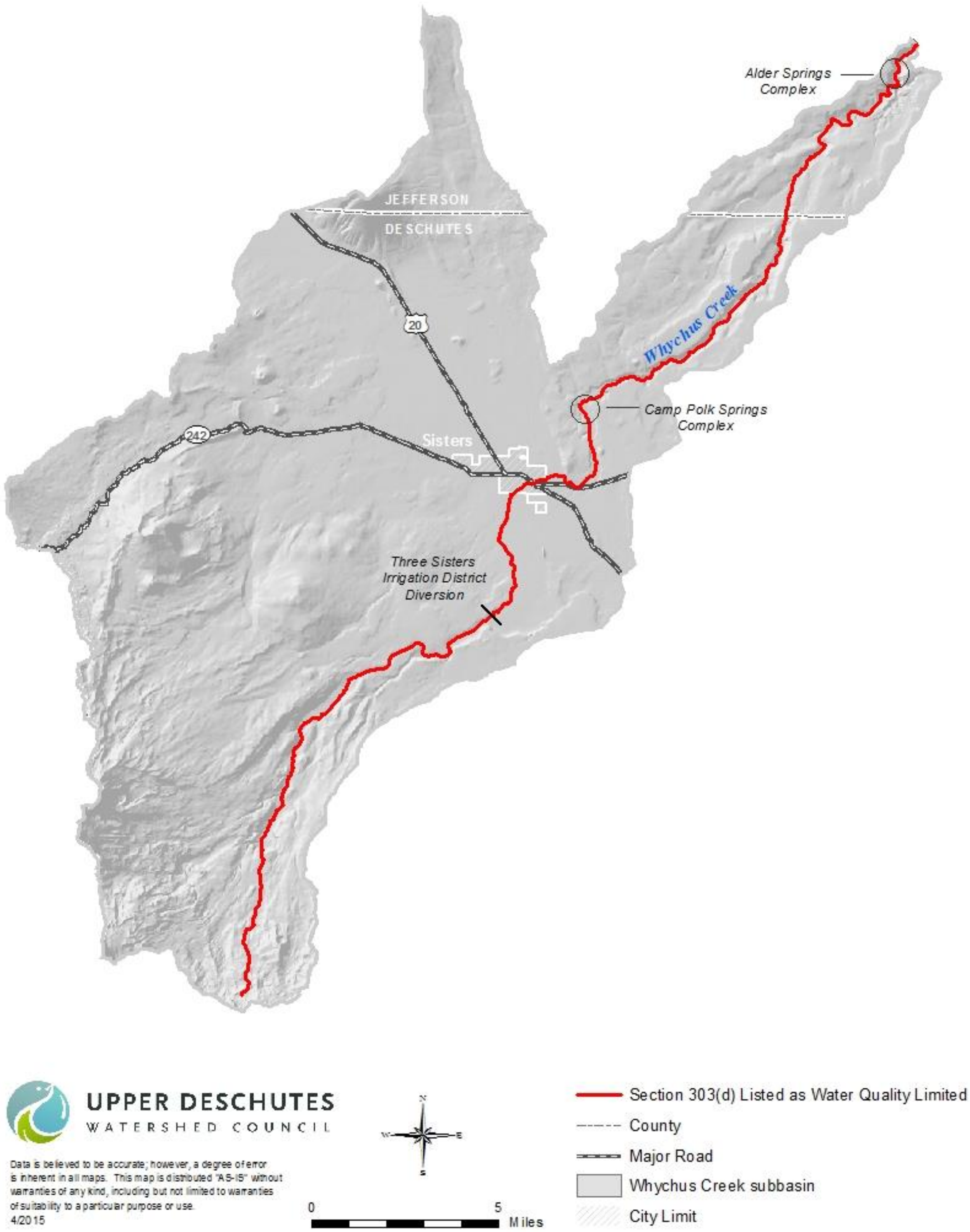


Figure 1.
Whychus Creek is listed as Water Quality Limited from river mile (RM) 0.0 to RM 40.3 under ODEQ's 2012 303(d) list. (ODEQ 2016)

Methods

Data collection

Stream Temperature Data

Beginning in 1995, UDWC and partners collected continuous temperature data annually at a subset of thirteen locations on Whychus Creek between river mile (RM) 38 and RM 0.25 (Figure 2, Appendix A). All temperature data used in analyses were collected by USFS, BLM, ODEQ, and UDWC. Coordinated monitoring efforts were conducted according to standard methods and protocols outlined in the ODEQ-approved UDWC Quality Assurance Project Plan (UDWC 2008a) and summarized in UDWC Water Quality Monitoring Program Standard Operating Procedures (UDWC 2008b).

In 2009 UDWC, Deschutes Land Trust (DLT), private landowners and other restoration partners reached an agreement to restore 1.9 miles of the historic meadow channel of Whychus Creek at Rimrock Ranch. The planned restoration will divert the creek from the existing channel into the meadow, and the UDWC monitoring station historically located on the existing channel will no longer be on the stream. To replace this monitoring location and generate pre-restoration data above and below the restoration project site UDWC established two new temperature monitoring stations, one upstream and one downstream of the planned restoration. As of 2009 UDWC discontinued temperature monitoring at the old Rimrock temperature monitoring station at WC 009.00 and began monitoring temperatures at the two new locations. Site names assigned to the two new sites are based on distance from the original WC 009.00 site. Although the downstream site is 0.7 mi from WC 009.00, another site had already been designated as WC 008.25. We accordingly designated the downstream Rimrock site as WC 008.50, the next closest quarter-mile increment.

Stream Flow Data

We obtained average daily stream flow (QD) data for Whychus Creek from Oregon Water Resources Department (OWRD) gage 14076050 at the City of Sisters (OWRD 2015). This gage is located downstream from the Three Sisters Irrigation District diversion and other major irrigation diversions. We use data collected at this gage from 2000 to 2015 in this report, including some data considered by OWRD to be provisional and subject to change.

Air Temperature Data

We obtained daily maximum air temperature data from the Colgate, Oregon Western Regional Climate Center (WRCC 2015) RAWS station (44° 18’ 57”, 121° 36’ 20”), the closest RAWS station to Whychus Creek.

Data analysis

Stream Temperature Status

We used the Oregon Department of Environmental Quality (ODEQ) Hydrostat Simple spreadsheet (ODEQ, 2010) to calculate the seven day moving average maximum () temperature, the statistic used by the State of Oregon to evaluate stream temperature. The State of Oregon water temperature standard for salmon and trout rearing and migration identifies a threshold of 18°C/64°F (OAR 340-041-0028). Because steelhead spawning season has yet to be identified for Whychus Creek, we reference the January 1 – May 15 spawning season identified for the Lower Deschutes sub-basin for evaluation of temperature relative to the 13˚C state standard for steelhead and salmon spawning. Chinook salmon spawning in Whychus Creek is anticipated to occur from late August through early October with incubation occurring through March or April (personal communication, B. Spateholts, February 15, 2015), earlier than the October 15 – May 15 and October 15 – June 15 spawning and incubation dates designated for the lower Deschutes.

We evaluated April 1 – October 31 temperatures from 2001-2015 in relation to the state standard of 18°C and the 13˚C state standard for steelhead and salmon spawning to describe changes in temperature in Whychus Creek since 2001 and to assess progress toward the 18°C state standard for salmonid rearing and migration, and identified 2015 flows at which stream temperatures exceeded the rearing and migration and spawning criteria. We calculated the percent of data days exceeding the 18°C and 13°C standards as the sum of the percentage of data days exceeding 13°C between April 1 and May 15 and the percentage of data days exceeding 18°C from May 16 to October 31. Because the Chinook salmon spawning dates anticipated for Whychus Creek are different than the designated use on the lower Deschutes, we evaluated September dates exceeding 13˚C separately from the percent of days exceeding the two standards.

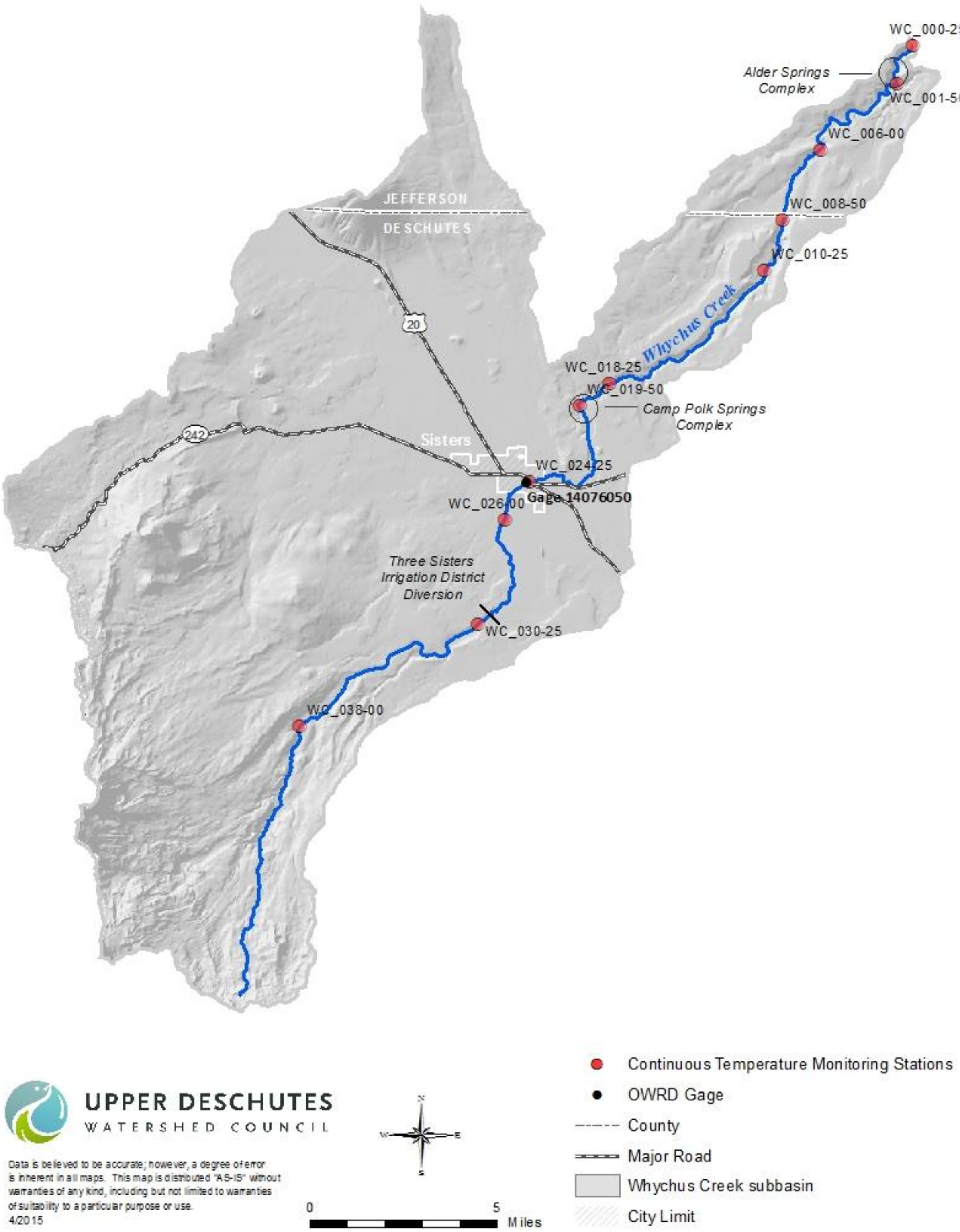


Figure 2.
Continuous temperature monitoring stations monitored in 2014, and OWRD Gage 14076050 at Sisters City Park, on Whychus Creek.

Target Stream flow

We used regressions of stream temperature, stream flow, and air temperature data to 1) quantify the effects of stream flow and air temperature on stream temperature, and 2) to calculate stream flows predicted to produce the 18°C rearing temperature standard and 13°C steelhead spawning temperature standard at key monitoring sites.

While the use of air temperature to predict stream temperature has been the subject of debate within the scientific community, we included air temperature in regressions on the basis of an extensive body of scientific literature supporting its application for this purpose. Air temperature has been shown to be a useful proxy for heat energy transfer from the atmosphere to water by long-wave radiation and sensible heat transfer (Webb and Zhang 1997; Mohseni and Stefan 1999), and multiple studies have used air temperature to accurately predict stream temperature variation (e.g. Webb et al. 2003; Mohseni et al. 2003; Morrill et al. 2005; Carlson et al. 2015).

We used temperature data for each year and site included in the analysis with corresponding stream flow data from the OWRD gage at Sisters City Park and air temperature data from the Colgate, OR Western Regional Climate Center RAWS station (WRCC 2015). We restricted data included in each regression to a one-month (30-day) interval to reduce the effect of intra-annual seasonal variation in the analysis (Helsel and Hirsch 2002). To calculate stream flows required to produce 18°C we evaluated July stream temperature data from WC 024.25 and WC 006.00. We selected July as the historically hottest month for stream temperature in Whychus Creek (UDWC unpublished data). Temperature data from WC 024.25 represent stream conditions immediately below major irrigation diversions; data from WC 0006.00 represent the historically worst temperature conditions on the creek, and thus the location that is both most critically in need of and also stands to benefit the most from stream flow restoration. To calculate stream flows required to produce 13°C during the January 1 – May 15 spawning season we evaluated April stream temperature data from WC 006.00. We selected April as the month during which stream temperature most often begins to exceed the 13°C steelhead spawning standard, and evaluated the relationship between stream temperature and stream flow at WC 006.00 as the site which typically represents the highest stream temperatures. To calculate stream flows required to produce 13°C during the late August to early October period during which Chinook salmon spawning is anticipated to occur in Whychus Creek we evaluated September stream temperature data from WC 006.00. We selected September as the month encompassing the majority of dates during which Chinook salmon are anticipated to spawn. For data for each month we evaluated the effect of air temperature on stream temperature to account for variation in stream temperature not explained by stream flow.

For each site and month we included all dates for which stream temperature, stream flow, and air temperature data were available. We used R open source statistical software (R Core Team, 2015) to perform linear, quadratic, and cubic regressions for each site: 1) with each of two flow metrics (average daily flow and the natural logarithm of average daily flow); and 2) with each of two air temperature metrics (daily maximum and three-day moving average maximum; 3DAir) for a total of twelve models (Table 2), to evaluate which metrics and models best described the data. The resulting equations represent the relationship between flow and temperature and can be used to estimate temperature values for the specified locations, within the evaluated time period, and within the range of flows observed.

Table 2. Twelve regression models evaluated for Whychus Creek at WC 024.25 and WC 006.00.

	Regression Model
1.	7DADM ~ QD
2.	7DADM ~ QD + $(QD)^2$
3.	7DADM ~ QD + $(QD)^2$ + $(QD)^3$
4.	7DADM ~ Ln QD
5.	7DADM ~ Ln QD + $(\text{Ln QD})^2$
6.	7DADM ~ Ln QD + $(\text{Ln QD})^2$ + $(\text{Ln QD})^3$
7.	7DADM ~ Air
8.	7DADM ~ Air + $(\text{Air})^2$
9.	7DADM ~ Air + $(\text{Air})^2$ + $(\text{Air})^3$
10.	7DADM ~ 3DAir
11.	7DADM ~ 3DAir + $(\text{3DAir})^2$
12.	7DADM ~ 3DAir + $(\text{3DAir})^2$ + $(\text{3DAir})^3$

We used the extractAIC function in R to generate Akaike Information Criterion (AIC) values for each regression model. AIC values rank models relative to each other on the basis of goodness of fit and number of parameters, with values decreasing as models improve; the lowest value indicates the best model. A difference of two or more between AIC values for two models denotes a statistically better model. For each site we evaluated R-squared (R^2), residual standard error (S), and AIC values to select the model that resulted in the best fit to the observed data; we evaluated residuals plots and normal probability plots for normality of residuals for the best model.

Using the best regression model for each site for July and April, we used R to calculate the predicted temperature and 95% prediction interval for all flows within the observed range (Appendix A). The 95% prediction interval (PI) is calculated as:

$$\hat{y}^*_i \pm T_{df=n-2,\alpha/2} * SE(\hat{y}^*_i \mid x_o)$$

where T is the $1-\alpha/2^{th}$ percentile of a T distribution with n-2 degrees of freedom.

For July data, we compared the resulting 2000-2015 temperature-flow regressions and predicted temperatures at given flows for each site to Heat Source model scenarios for the same locations on Whychus Creek (Watershed Sciences and MaxDepth Aquatics 2008). Because available Heat Source scenarios assume 33 cfs at WC 024.25 and 62 cfs at WC 006.00, we compared 2000-2015 predicted temperatures to Heat Source estimates for these flows.

Results

Temperature status

Seven-day moving average maximum (7DADM) temperatures exceeded the 18°C state standard for trout and salmon rearing and migration at eight locations between rm 6.0 and rm 26.0, by up to °C, and exceeded the January 1 – May 15 13°C state standard for steelhead spawning at nine locations between rm 1.5 and rm 26 by up to 3.2°C, in 2015, (Figure 3), supporting the existing State of Oregon Section 303(d) listing of Whychus Creek as water quality limited.

Percent of data days exceeding 18°C between April and October and 13°C between January 1 and May 15 at WC 006.00 approximately represent the maximum amount of time annually during which stream conditions are unsuitable for rearing trout and for spawning steelhead trout, respectively, in Whychus Creek; conversely, the percent of days meeting 18°C, 13°C between January 1 and May 15, represents the amount of time during which stream conditions are suitable to support rearing and spawning fish. The number of days for which stream temperature data were available ranged from 113 days in 2009 to 207 days in 2013.

Stream temperature at WC 006.00 exceeded the applicable temperature standards for 58% of data days (119 days) in 2015 (Figure 4), higher than in the previous six years (data for 2008 are unavailable), and higher than in all but five of the thirteen total years for which data are available. Temperatures at this site met the applicable standard, providing suitable conditions for rearing trout and spawning steelhead, for 42% of data days (85 days) in 2015. Temperatures remained below 20°C for 62% of 2015 data days (125 days), exceeded 20°C and remained below 24°C for 37% of data days (76 days), and exceeded 24°C for 1% of data days (3 days) in 2015. Stream temperature exceeded 13°C between April 15 and May 15 at Sisters City Park flows of 19 to 27; stream temperature exceeded 18°C between May 25 and August 30 at Sisters City Park flows of 14 to 56 cfs.

Stream temperature at WC 006.00 exceeded 13°C for the entire month of September in 2015. Temperatures exceeded 13°C for all September days for which data were available in seven of twelve years, including 2015; in the remaining five years temperatures exceeded 13°C for 70-97% of data days. September stream flow at Sisters City Park over twelve years from 2000-2015 ranged from 1 to 43 cfs, with a median flow of 16 cfs. Anomalously high flows between 105 cfs and 400 cfs occurred from September 28-30, 2013.

July stream temperatures at Sisters City Park and Road 6360 in 2015 were the highest observed since 2007[1], corresponding to the lowest July flows since 2007 (Figure 5). Median July temperature trends at Sisters City Park and Road 6360 closely tracked July median stream flow.

[1]July temperatures may have been higher in 2009 than in 2015 but insufficient data are available to evaluate 2009 temperatures.

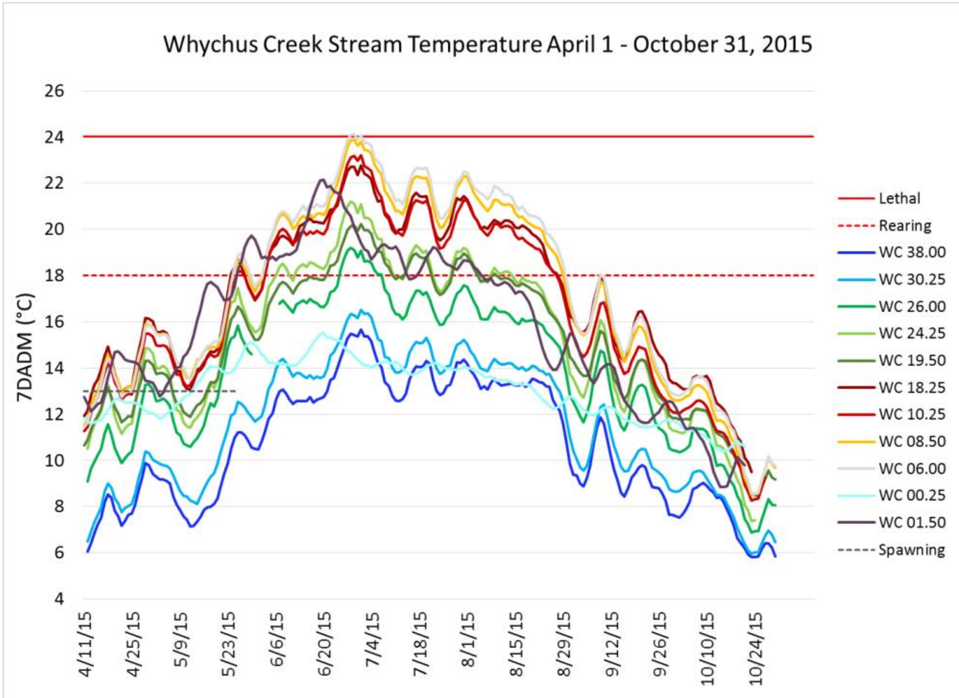


Figure 3. 2015 temperatures at eleven Whychus Creek monitoring sites. Stream temperature exceeded the 18˚C rearing standard at eight sites in 2015, from rm 1.5 (WC 001.50) to rm 26.00 (WC 026.00), and exceeded the January 1-May 15 13˚C steelhead spawning standard at nine sites from rm 0.25 (WC 000.25) to rm 26.

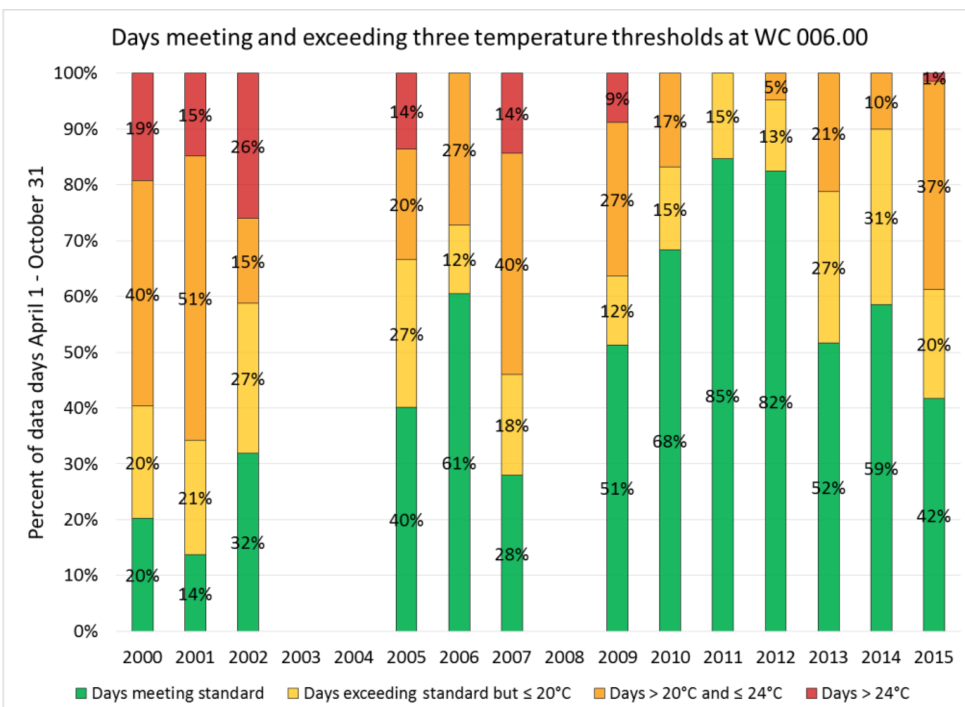
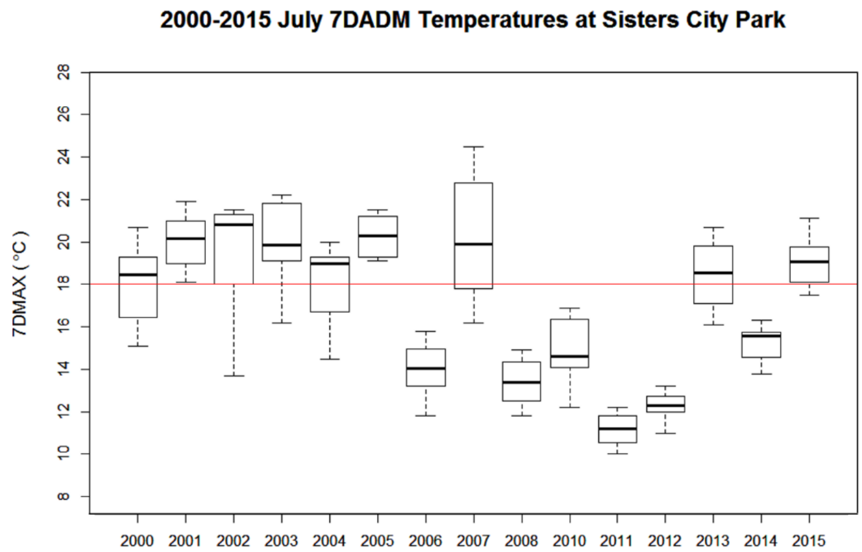


Figure 4. Percent of data days meeting and exceeding three temperature thresholds at WC 006.00. Stream temperatures exceeded the spawning (13˚C) or rearing (18˚C) state standard for 42% of days in 2015; temperatures remained below 20 ˚C for an additional 20% of days in 2015, were above 20 ˚C but below the lethal 24 ˚C threshold for 37% of days, and exceeded 24 ˚C

a



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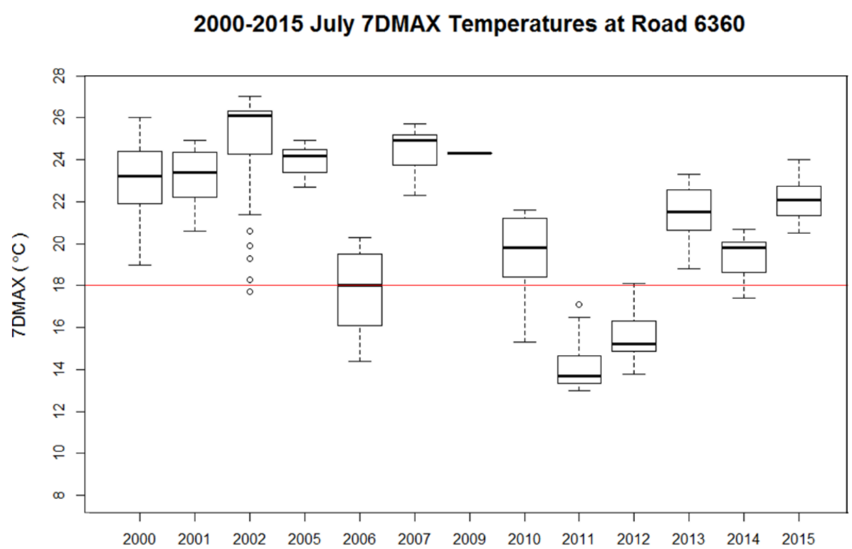

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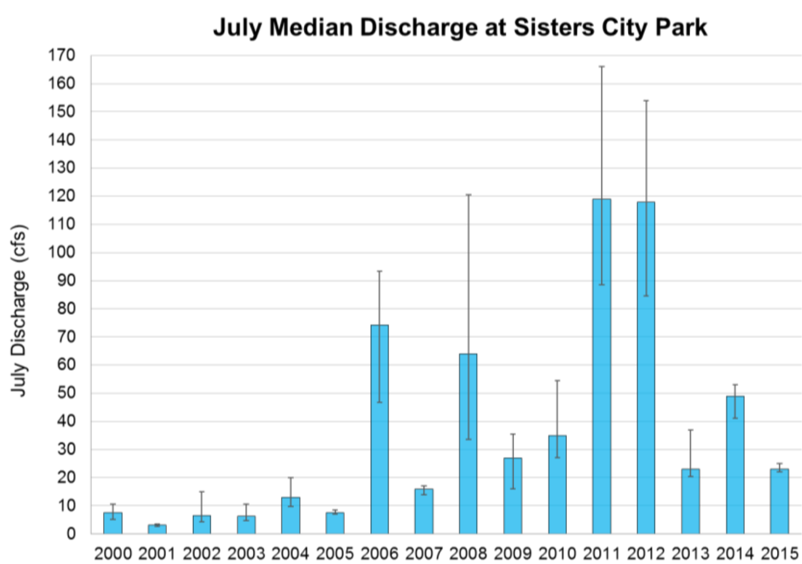


Figure 5. 2001-2015 July Median Stream Temperature and Stream Flow

July stream temperatures at a) Sisters City Park (WC 024.25) and b) Road 6360 (WC 006.00) correspond closely to c) stream flow at Sisters City Park.

Target stream flow

Rearing and migration temperature standard

Temperature records were available from WC 024.25 and from WC 006.00 for July dates from 2000 through 2015 at Sisters City Park flows from 2 to 201 cfs (Table 3). The cubic regression of stream temperature on the natural log of average daily flow (7DADM ~ LnQD + $(LnQD)^2$ + $(LnQD)^3$) performed best of the twelve regression models for both sites (Table 4). Using this model, stream flow explained 79% and 83% of the variation in stream temperature in July at WC 024.25 and at WC 006.00, respectively ($R^2 = 0.79$; $R^2 = 0.83$). The linear regression of stream temperature on three-day moving average air temperature performed the best of the six air temperature models for both sites, explaining 20% and 21% of the variation in stream temperature at WC 024.25 and WC 006.00 ($R^2 = 0.20$; $R^2 = 0.21$).

Temperatures calculated from the July WC 024.25 cubic regression model suggest that 22 cfs was the minimum stream flow resulting in a mean temperature at or below 18°C (± 3°C) given temperatures observed from July 2000-2015 at Sisters City Park (Appendix A); allowing for the 3°C prediction interval, 51 cfs is predicted to result in an upper limit stream temperature of 18°C at Sisters City Park. The existing 33 cfs restoration target predicts a mean temperature of 16.6°C ± 3°C at this site. Although direct comparison to Heat Source model predictions is not possible because Heat Source uses the seven day moving average maximum temperature, a daily statistic, and we use the mean seven day moving average maximum temperature for July, a monthly statistic, our 2000-2015 estimate for Sisters City Park is substantially (1.6°C) higher than the 2008 Heat Source model estimate of 15°C ± 1°C at 33 cfs at the ODFW gage at Sisters City Park (Watershed Sciences and MaxDepth Aquatics 2008).

The cubic regression of 2000-2015 temperature and the natural log of flow at Road 6360 (WC 0006.00) estimates 63 cfs to be the minimum stream flow that will achieve a mean temperature of 18.0°C ± 3°C. According to this model the target stream flow of 33 cfs below Indian Ford Creek is projected to produce a mean temperature of 20.7°C ± 3°C at Road 6360. The 2000-2015 cubic regression model estimate of 18.1°C ± 3°C at 62 cfs is slightly lower than the Heat Source model estimate of 18.5°C ± 1°C at 62 cfs at Road 6360.

Steelhead and salmon spawning standard

Temperature records were available from WC 006.00 for April dates from 2001 through 2015 at Sisters City Park flows from 2 to 128 cfs (Table 3). The linear regression of stream temperature on the natural log of average daily flow (7DADM ~ LnQD) performed best of the twelve regression models (Table 4). Stream flow explained only 53% of the variation in stream temperature in April ($R^2 = 0.53$). The linear regression of stream temperature on three-day moving average air temperature performed best of the six air temperature models, explaining 33% of the variation in stream temperature ($R^2 = 0.33$).

Because the April regression models explained relatively little of the variation in stream temperature, we used the same methods to evaluate the same relationships for May stream temperature, stream flow, and air temperature data. May regressions explained less of the variation in stream temperature than April regressions. The cubic regression of stream temperature on average daily flow (7DADM ~ QD + $(QD)^2$ + $(QD)^3$) performed best of the twelve models for May and explained 26% ($R^2 = 0.26$) of the variation in stream flow; the best model for air temperature, the quadratic regression of stream temperature on three day moving average air temperature (7DADM ~ 3DAir + $(3DAir)^2$), explained 20% of the variation in stream temperature ($R^2 = 0.26$).

Multiple regression of May stream temperature on the natural log of average daily flow and the three day moving average air temperature (7DADM ~ LnQD + 3DAir) performed best of twelve linear and multiple regression models in a temperature assessment analysis conducted for the Upper Deschutes Basin Study (UDWC 2016), explaining 74% of the variation in stream temperature. To attempt to explain a greater proportion of the variation in stream temperature in May we incorporated 2015 data into the Basin Study 7DADM ~ LnQD + 3DAir regression. The resulting model explained 47% (R^2 = 0.47) of the variation in stream temperature. We verified the 2001-2015 multiple regression model by using the same code in R used in the Basin Study assessment.

We used the April linear regression of stream temperature on the natural log of average daily flow (7DADM ~ LnQD), which explained the greatest proportion of variation in stream temperature of the April and May regression models, to calculate temperatures at the range of April flows. This model predicts a mean stream temperature of 13°C±3.2°C (a range encompassing 9.8°C – 16.2°C) to occur at 18 cfs at Sisters City Park. The state instream water right and DRC stream flow target of 33 cfs resulted in 11.7°C±3.2°C (8.5°C – 14.9°C). Eighty-one cfs were required at WC 024.25 to produce 13.0°C (mean 9.8°C) as the upper limit of the prediction interval at WC 006.00 in April.

Temperature records were available from WC 006.00 for September dates from 2000 through 2015 at Sisters City Park flows from 1 to 400 cfs. The best-performing model of the twelve models, the quadratic regression of stream temperature on the three-day moving average maximum air temperature, explained only 31% of the variation in stream temperature (R^2 = 0.31;Table 4). The best-performing stream flow model, the quadratic regression of the natural log of average daily flow, explained even less at 26% (R^2 = 0.26). To attempt to fit a model that incorporated stream flow while explaining greater than 26% of the variation in stream temperature in September, we incorporated 2015 data into the Basin Study 7DADM ~ LnQD + 3DAir multiple linear regression. The resulting model explained 52% (R^2 = 0.52) of the variation in stream temperature. This model predicted a mean temperature of 15°C±3.0°C (11.9°C – 18.0°C) at the natural log of 33 cfs (3.4965 LnQD). Because data were not available for flows between 43 and 100 cfs, we did not calculate a predicted temperature for higher flows.

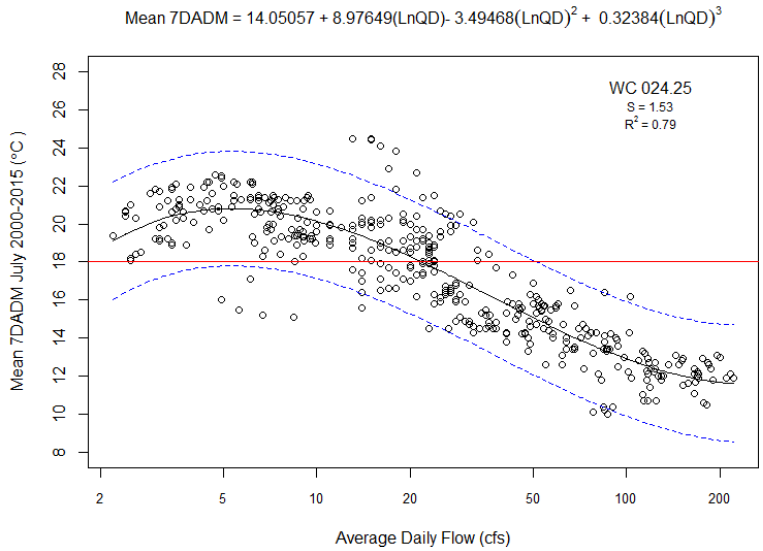
Table 3. Years for which data are available and which are represented in regression analyses. The number of days for which data are available for any given month varies.

Site and Month	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015
WC 024.00																
July	x	x	x	x	x	x	x	x	x		x	x	x	x	x	x
WC 006.00																
April		x	x			x				x	x		x	x	x	x
July	x	x	x			x	x	x		x	x	x	x	x	x	x
September	x	x				x	x	x		x	x	x	x	x	x	x

Table 4. A cubic regression model provided the best fit to July 2000-2015 temperature-flow data for both WC 024.25 and WC 0006.00 data. A linear regression model provided the best fit to April 2000-2015 temperature-flow data. Temperatures calculated using the corresponding regression equations are expected to be the most accurate of the regression models evaluated.

Regression Model	Intercept	Coefficient 1	Coefficient 2	Coefficient 3	n	df	R^2	S	AIC value
July - WC 024.25									
7DADM ~ LnQD + $(LnQD)^2$ + $(LnQD)^3$	**14.05057**	**8.97649**	**-3.49468**	**0.32384**	**427**	**423**	**0.793**	**1.525**	**364**
7DADM ~ QD + $(QD)^2$ + $(QD)^3$	21.39	-0.1702	0.001016	-0.000002	427	423	0.791	1.532	368
7DADM ~ QD + $(QD)^2$	21.01	-0.1343	0.00047	--	427	424	0.78	1.558	382
July - WC 006.00									
7DADM ~ LnQD + $(LnQD)^2$ + $(LnQD)^3$	**17.24498**	**9.13025**	**-3.23634**	**0.26028**	**342**	**338**	**0.829**	**1.51**	**287**
7DADM ~ QD + $(QD)^2$	25.07	-0.1381	0.00044	--	342	339	0.826	1.527	293
7DADM ~ QD + $(QD)^2$ + $(QD)^3$	25.04	-0.1347	0.0003845	0.00000021	342	338	0.826	1.529	295
April - WC 006.00									
7DADM ~ LnQD	**19.0586**	**-2.1061**	--	--	**136**	**134**	**0.528**	**1.614**	**132**
7DADM ~ QD + $(QD)^2$ + $(QD)^3$	21.7865	-5.8927	1.5044	-0.1801	136	132	0.534	1.615	134
7DADM ~ QD + $(QD)^2$	18.38184	-1.63797	-0.07538	--	136	133	0.528	1.618	134
September - WC 006.00									
7DADM ~ LnQD + 3DAir	**13.55792**	**-1.31134**	**0.2328**	**--**	**279**	**276**	**0.52**	**1.54**	**311**
7DADM ~ 3DAir + $(3DAir)^2$ + $(3DAir)^3$	-7.63511	2.429945	-0.08733	0.001122	279	275	0.31	1.85	347
7DADM ~ 3DAir + $(3DAir)^2$	6.280263	0.557711	-0.00659	--	279	276	0.30	1.86	348

a



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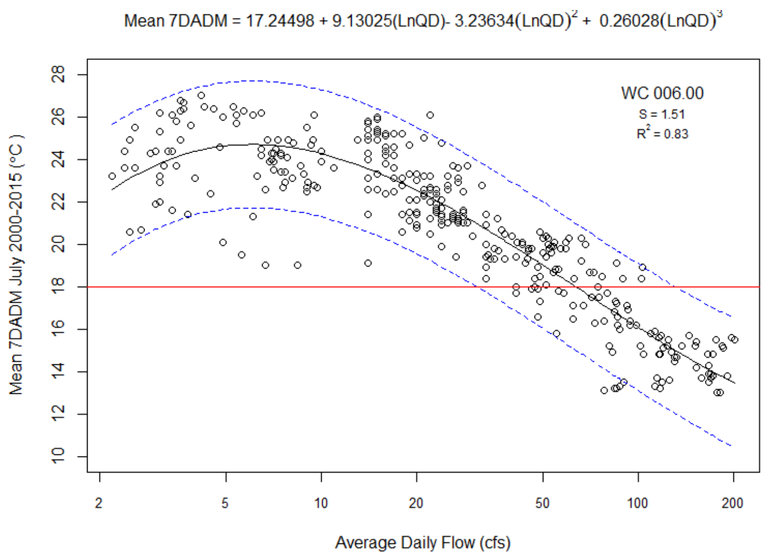

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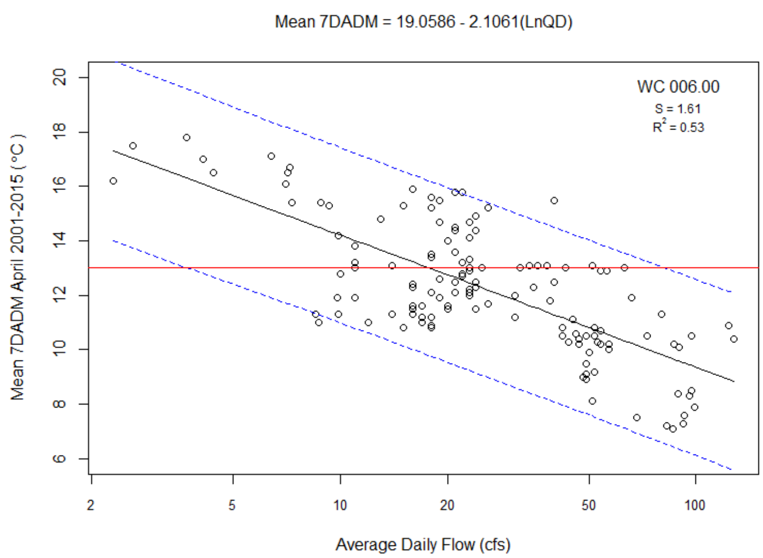

Figure 8. Temperature-Flow Regression Models

Regression models fitted to temperature-flow data demonstrate reduced temperatures at higher flows and describe the relationship between temperature and flow observed a) during July 2000-2015 at WC 024.25, b) during July 2000-2015 at WC 006.00, and c) during April 2001-2015 at WC 006.00.

Discussion

Temperature status and trend

stream temperature exceeded the state standard for trout rearing and migration in 2015, supporting the ODEQ 2012 303(d) Category 5 listing of Whychus Creek as water quality limited (ODEQ 2014). Stream temperatures exceeding the 18˚C standard over a prolonged duration suggest temperature conditions compromised habitat suitability for rearing and migrating trout and salmon in Whychus Creek from Sisters City Park (WC 024.25) to rm 1.5 (WC 001.50) in 2015. Seven day moving average maximum temperatures above 13 ˚C for the majority of January 1-May 15 spawning season data days as well as for the majority of September data days downstream of WC 18.25, also indicate marginal spawning conditions for steelhead and Chinook salmon. Stream temperature reached the 24˚C lethal threshold for three days at WC 006.00 in 2015.

Stream temperature exceeded the state standard for fewer days in 2015 than in five years from 2000 to 2002, in 2005, and in 2007, showing a sustained improvement over early years of stream flow restoration (data from WC 006.00 are not available for 2003 and 2004). This year simultaneously marks the worst of three successive years of lower stream flows and higher stream temperatures, reversing the trend of higher flows and lower temperature observed from 2009 through 2012. Notably, despite 2015 being the hottest and driest year on record for the Deschutes Basin (NOAA 2016; NRCS 2015), July median flow at Sisters City Park was more than 5 cfs higher than in 2007, the year characterized by the previous lowest July median flow at Sisters City Park, and 2015 average daily flows remained above 20 cfs throughout July, likely buffering Whychus Creek stream temperatures from even higher rates of warming. Relatively high July 2015 flows compared to natural July 2015 flows (Appendix B) demonstrate both progress in stream flow restoration and a significant commitment by TSID irrigation district management to practices that maintain a minimum of 20 cfs in Whychus Creek downstream of the TSID diversion.

Regression of temperature and flow data as well as comparison of median monthly temperature and stream flow data and mean 7DADM temperatures for given flow levels show stream temperatures decreasing as flows increase. Stream flow restoration has increased the minimum flow delivered instream, resulting in higher July median flows that reflect consistently higher average daily flows, which in turn correspond to lower observed temperatures.

Target stream flow

The state water right for Whychus Creek protects 20 cfs instream above Indian Ford Creek, between RM 20 and RM 21, and 33 cfs downstream of Indian Ford Creek. Because no additional flows enter Whychus Creek between the headwaters and Indian Ford Creek, DRC established a stream flow restoration target of 33 cfs for the entire length of the creek from headwaters to mouth. July regression results from Road 6360 (WC 0006.00) 2000-2015 temperature and flow data indicate a minimum flow of 63 cfs is necessary to achieve stream temperatures of 18°C±3°C at this site. According to this model the target stream flow of 33 cfs below Indian Ford Creek is projected to produce a mean temperature of 20.7°C ± 3°C at Road 6360, above the 18°C state standard and the 20°C threshold shown to increase mortality in trout (Runge *et al* 2008); the highest temperature predicted at this flow, 23.7°C, just misses the lethal temperature threshold for trout of 24°C.

Regression of April stream temperature and flow data suggests the 33 cfs DRC stream flow restoration target will result in stream temperatures between 8.5 and 14.9°C at WC 006.00, encompassing and exceeding the 13°C spawning threshold (predicted mean 7DADM = 11.7°C±3.2°C). This result suggests 33 cfs will support suitable steelhead spawning temperatures some of the time; the influence of air temperature on stream temperature in April, explaining 33% of the variability in stream temperature during this month, suggests air temperature will determine whether stream temperature meets or exceeds the spawning criteria at 33 cfs. Although 18 cfs is predicted to result in a mean 13°C±3.2 stream temperature, temperatures exceeding that criteria at flows of 20 cfs and higher support the need for 33 cfs or higher in April and May.

Regression of September stream temperature, stream flow, and air temperature data suggests 33 cfs will result in stream temperatures between 11.9°C and 18.0°C (predicted mean 7DADM = 15.0°C±3.0°C) at the median September air temperature of 25.7°C (78.3°F), also encompassing, and exceeding by a greater amount than in April, the 13°C spawning threshold during anticipated Chinook salmon spawning. This result suggests 33 cfs will inconsistently support suitable Chinook salmon spawning temperature, depending in large part on air temperature. The lack of September flow records above 43 cfs at Sisters City Park limits our ability to make predictions about what September flows may provide conditions that support the 13°C spawning criteria for Chinook salmon.

These results clearly demonstrate the current state water right of 33 cfs is well below the stream flow necessary to meet state standards and provide suitable conditions for rearing and migrating native trout and salmon, and support the conclusion of previous regression models and Heat Source model results (Watershed Sciences and MaxDepth Aquatics 2008). In addition, minimum flows that on average have resulted in 18°C may not be sufficient to meet that threshold in hotter years given the influence of air temperature on stream temperature. Flows above 50 cfs are predicted to maintain July temperatures below the 22°C threshold at which trout have been shown to suffer severe effects of chronic sub-lethal temperatures and also maintain April temperatures below the 15°C threshold at which egg morality increases.

Conclusions

Stream flow restoration and TSID management practices have achieved some sustained improvements in reducing the magnitude and duration of high stream temperatures in Whychus Creek. In particular, July stream temperatures at Road 6360 (WC 006.00) have been consistently lower in the last six years (2010-2015) than in early years of stream flow restoration (2000-2005; 2007), as has the percent of days exceeding the 18°C state standard. These results suggest some improvement in the suitability of stream conditions in Whychus Creek for rearing trout during the irrigation season.

Regression analyses of empirical stream temperature and stream flow data substantiate Heat Source model results showing more than 60 cfs is required to meet 18°C on average in lower reaches of Whychus Creek in July; stream temperatures as high as 21°C are predicted to occur at 63 cfs, emphasizing the imperative need for 60 cfs as a minimum flow during July to reduce stream temperatures below the threshold at which trout experience chronic effects that result in mortality.

Although 60 cfs may not be a feasible restoration target given current land and water use in the Three Sisters Irrigation District, these data provide a benchmark for stream flow restoration and, importantly, show the 33 cfs state water right to be far short of the flows needed to meet the state temperature standard or provide suitable conditions for fish. Small gains in stream flow restoration that result in similarly small reductions in temperature are nonetheless likely to improve habitat conditions for some fish in some locations, for example by providing adequate flow for steelhead outmigration, increasing channel margin habitat by increasing channel width, and creating pools and cover for resident redband.

Our results show that higher stream flow achieved in part through stream flow restoration results in lower temperatures and better stream conditions for re-introduced salmon and trout, highlight the significant need for higher flows to achieve suitable conditions for salmon and trout in Whychus, and contribute to an improved understanding of temperature and flow that that we hope will support restoration partners in planning more ambitious stream flow restoration efforts on Whychus Creek.

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APPENDIX A Temperatures at given flows.

Whychus Creek at Sisters City Park (WC 024.25) predicted temperatures for July at flows from 2 to 200 cfs

Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)
2	18.7	3.1	57	14.6	3.0	112	12.6	3.0	167	11.9	3.0
3	20.1	3.0	58	14.6	3.0	113	12.6	3.0	168	11.9	3.0
4	20.6	3.0	59	14.5	3.0	114	12.6	3.0	169	11.9	3.0
5	20.8	3.0	60	14.4	3.0	115	12.6	3.0	170	11.8	3.0
6	20.8	3.0	61	14.4	3.0	116	12.5	3.0	171	11.8	3.0
7	20.7	3.0	62	14.3	3.0	117	12.5	3.0	172	11.8	3.0
8	20.5	3.0	63	14.3	3.0	118	12.5	3.0	173	11.8	3.0
9	20.3	3.0	64	14.2	3.0	119	12.5	3.0	174	11.8	3.0
10	20.1	3.0	65	14.2	3.0	120	12.5	3.0	175	11.8	3.0
11	19.9	3.0	66	14.1	3.0	121	12.4	3.0	176	11.8	3.0
12	19.7	3.0	67	14.1	3.0	122	12.4	3.0	177	11.8	3.0
13	19.5	3.0	68	14.0	3.0	123	12.4	3.0	178	11.8	3.0
14	19.4	3.0	69	14.0	3.0	124	12.4	3.0	179	11.8	3.0
15	19.2	3.0	70	13.9	3.0	125	12.4	3.0	180	11.8	3.0
16	19.0	3.0	71	13.9	3.0	126	12.4	3.0	181	11.8	3.0
17	18.8	3.0	72	13.9	3.0	127	12.3	3.0	182	11.8	3.0
18	18.6	3.0	73	13.8	3.0	128	12.3	3.0	183	11.8	3.0
19	18.5	3.0	74	13.8	3.0	129	12.3	3.0	184	11.8	3.0
20	18.3	3.0	75	13.7	3.0	130	12.3	3.0	185	11.7	3.0
21	18.1	3.0	76	13.7	3.0	131	12.3	3.0	186	11.7	3.0
22	18.0	3.0	77	13.6	3.0	132	12.3	3.0	187	11.7	3.0
23	17.8	3.0	78	13.6	3.0	133	12.2	3.0	188	11.7	3.0
24	17.7	3.0	79	13.6	3.0	134	12.2	3.0	189	11.7	3.0
25	17.5	3.0	80	13.5	3.0	135	12.2	3.0	190	11.7	3.1
26	17.4	3.0	81	13.5	3.0	136	12.2	3.0	191	11.7	3.1
27	17.3	3.0	82	13.5	3.0	137	12.2	3.0	192	11.7	3.1
28	17.1	3.0	83	13.4	3.0	138	12.2	3.0	193	11.7	3.1
29	17.0	3.0	84	13.4	3.0	139	12.2	3.0	194	11.7	3.1
30	16.9	3.0	85	13.4	3.0	140	12.1	3.0	195	11.7	3.1
31	16.8	3.0	86	13.3	3.0	141	12.1	3.0	196	11.7	3.1
32	16.7	3.0	87	13.3	3.0	142	12.1	3.0	197	11.7	3.1
33	16.6	3.0	88	13.3	3.0	143	12.1	3.0	198	11.7	3.1
34	16.4	3.0	89	13.2	3.0	144	12.1	3.0	199	11.7	3.1
35	16.3	3.0	90	13.2	3.0	145	12.1	3.0	200	11.7	3.1
36	16.2	3.0	91	13.2	3.0	146	12.1	3.0			
37	16.1	3.0	92	13.1	3.0	147	12.1	3.0			
38	16.0	3.0	93	13.1	3.0	148	12.1	3.0			
39	16.0	3.0	94	13.1	3.0	149	12.0	3.0			
40	15.9	3.0	95	13.0	3.0	150	12.0	3.0			
41	15.8	3.0	96	13.0	3.0	151	12.0	3.0			
42	15.7	3.0	97	13.0	3.0	152	12.0	3.0			
43	15.6	3.0	98	13.0	3.0	153	12.0	3.0			
44	15.5	3.0	99	12.9	3.0	154	12.0	3.0			
45	15.4	3.0	100	12.9	3.0	155	12.0	3.0			
46	15.4	3.0	101	12.9	3.0	156	12.0	3.0			
47	15.3	3.0	102	12.9	3.0	157	12.0	3.0			
48	15.2	3.0	103	12.8	3.0	158	11.9	3.0			
49	15.1	3.0	104	12.8	3.0	159	11.9	3.0			
50	15.1	3.0	105	12.8	3.0	160	11.9	3.0			
51	15.0	3.0	106	12.8	3.0	161	11.9	3.0			
52	14.9	3.0	107	12.7	3.0	162	11.9	3.0			
53	14.9	3.0	108	12.7	3.0	163	11.9	3.0			
54	14.8	3.0	109	12.7	3.0	164	11.9	3.0			
55	14.7	3.0	110	12.7	3.0	165	11.9	3.0			
56	14.7	3.0	111	12.6	3.0	166	11.9	3.0			

Whychus Creek at Road 6360 (WC 006.00) predicted temperatures for July at flows from 2 to 201 cfs

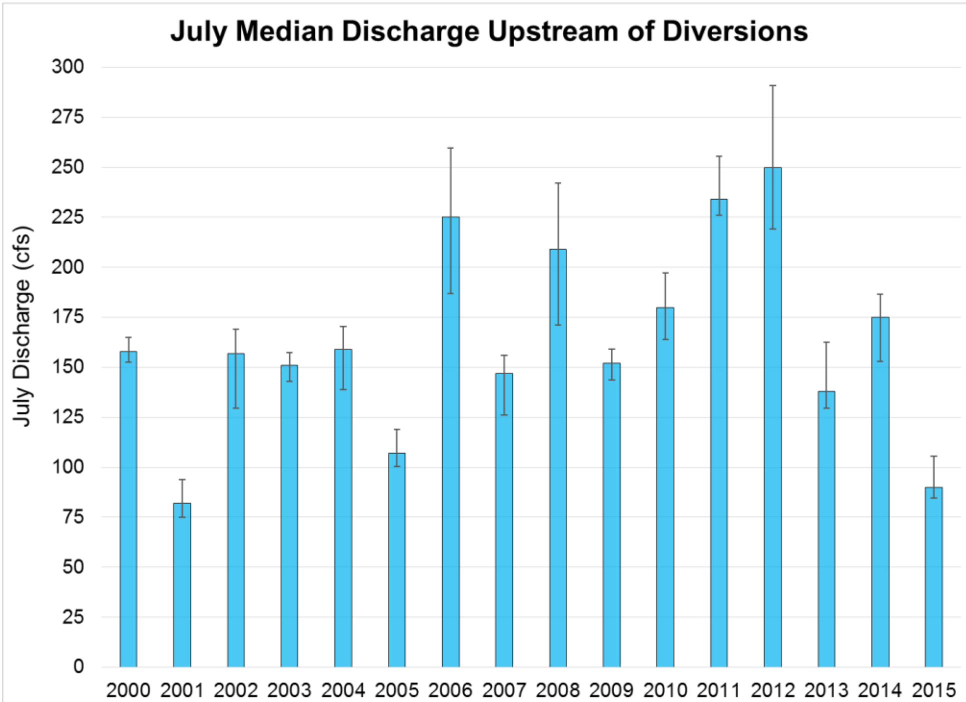
Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)
2	22.0	3.1	57	18.4	3.0	112	15.6	3.0	167	14.1	3.0
3	23.6	3.0	58	18.3	3.0	113	15.6	3.0	168	14.1	3.0
4	24.3	3.0	59	18.3	3.0	114	15.5	3.0	169	14.1	3.0
5	24.6	3.0	60	18.2	3.0	115	15.5	3.0	170	14.1	3.0
6	24.7	3.0	61	18.1	3.0	116	15.5	3.0	171	14.1	3.0
7	24.6	3.0	62	18.0	3.0	117	15.4	3.0	172	14.0	3.0
8	24.5	3.0	63	18.0	3.0	118	15.4	3.0	173	14.0	3.0
9	24.4	3.0	64	17.9	3.0	119	15.4	3.0	174	14.0	3.0
10	24.2	3.0	65	17.8	3.0	120	15.3	3.0	175	14.0	3.0
11	24.1	3.0	66	17.8	3.0	121	15.3	3.0	176	14.0	3.0
12	23.9	3.0	67	17.7	3.0	122	15.3	3.0	177	13.9	3.0
13	23.7	3.0	68	17.7	3.0	123	15.2	3.0	178	13.9	3.0
14	23.5	3.0	69	17.6	3.0	124	15.2	3.0	179	13.9	3.0
15	23.3	3.0	70	17.5	3.0	125	15.2	3.0	180	13.9	3.0
16	23.2	3.0	71	17.5	3.0	126	15.2	3.0	181	13.9	3.0
17	23.0	3.0	72	17.4	3.0	127	15.1	3.0	182	13.9	3.0
18	22.8	3.0	73	17.4	3.0	128	15.1	3.0	183	13.8	3.0
19	22.6	3.0	74	17.3	3.0	129	15.1	3.0	184	13.8	3.1
20	22.5	3.0	75	17.2	3.0	130	15.0	3.0	185	13.8	3.1
21	22.3	3.0	76	17.2	3.0	131	15.0	3.0	186	13.8	3.1
22	22.2	3.0	77	17.1	3.0	132	15.0	3.0	187	13.8	3.1
23	22.0	3.0	78	17.1	3.0	133	15.0	3.0	188	13.8	3.1
24	21.9	3.0	79	17.0	3.0	134	14.9	3.0	189	13.7	3.1
25	21.7	3.0	80	17.0	3.0	135	14.9	3.0	190	13.7	3.1
26	21.6	3.0	81	16.9	3.0	136	14.9	3.0	191	13.7	3.1
27	21.4	3.0	82	16.9	3.0	137	14.8	3.0	192	13.7	3.1
28	21.3	3.0	83	16.8	3.0	138	14.8	3.0	193	13.7	3.1
29	21.2	3.0	84	16.8	3.0	139	14.8	3.0	194	13.7	3.1
30	21.0	3.0	85	16.7	3.0	140	14.8	3.0	195	13.6	3.1
31	20.9	3.0	86	16.7	3.0	141	14.7	3.0	196	13.6	3.1
32	20.8	3.0	87	16.6	3.0	142	14.7	3.0	197	13.6	3.1
33	20.7	3.0	88	16.6	3.0	143	14.7	3.0	198	13.6	3.1
34	20.5	3.0	89	16.5	3.0	144	14.7	3.0	199	13.6	3.1
35	20.4	3.0	90	16.5	3.0	145	14.6	3.0	200	13.6	3.1
36	20.3	3.0	91	16.4	3.0	146	14.6	3.0	201	13.5	3.1
37	20.2	3.0	92	16.4	3.0	147	14.6	3.0			
38	20.1	3.0	93	16.4	3.0	148	14.6	3.0			
39	20.0	3.0	94	16.3	3.0	149	14.5	3.0			
40	19.9	3.0	95	16.3	3.0	150	14.5	3.0			
41	19.8	3.0	96	16.2	3.0	151	14.5	3.0			
42	19.7	3.0	97	16.2	3.0	152	14.5	3.0			
43	19.6	3.0	98	16.1	3.0	153	14.4	3.0			
44	19.5	3.0	99	16.1	3.0	154	14.4	3.0			
45	19.4	3.0	100	16.1	3.0	155	14.4	3.0			
46	19.3	3.0	101	16.0	3.0	156	14.4	3.0			
47	19.2	3.0	102	16.0	3.0	157	14.4	3.0			
48	19.1	3.0	103	15.9	3.0	158	14.3	3.0			
49	19.0	3.0	104	15.9	3.0	159	14.3	3.0			
50	19.0	3.0	105	15.9	3.0	160	14.3	3.0			
51	18.9	3.0	106	15.8	3.0	161	14.3	3.0			
52	18.8	3.0	107	15.8	3.0	162	14.2	3.0			
53	18.7	3.0	108	15.8	3.0	163	14.2	3.0			
54	18.6	3.0	109	15.7	3.0	164	14.2	3.0			
55	18.6	3.0	110	15.7	3.0	165	14.2	3.0			
56	18.5	3.0	111	15.6	3.0	166	14.2	3.0			

Whychus Creek at Road 6360 (WC 006.00) predicted temperatures for April at flows from 2 to 128 cfs

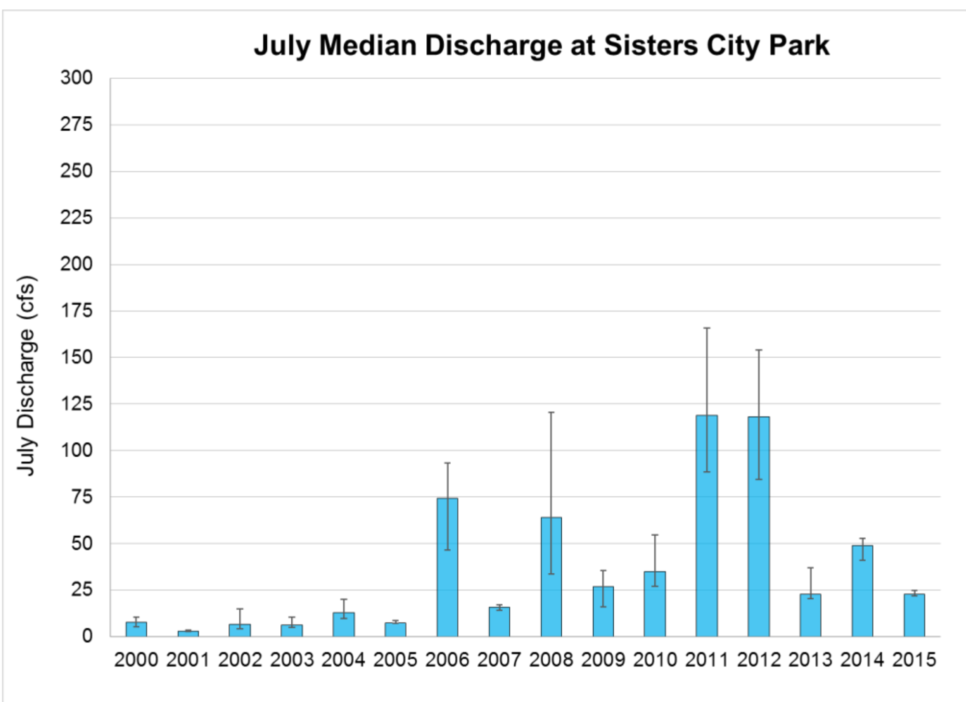
Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)
2	17.6	3.3	57	10.5	3.2	112	9.1	3.2			
3	16.7	3.3	58	10.5	3.2	113	9.1	3.2			
4	16.1	3.3	59	10.5	3.2	114	9.1	3.2			
5	15.7	3.3	60	10.4	3.2	115	9.1	3.2			
6	15.3	3.2	61	10.4	3.2	116	9.0	3.2			
7	15.0	3.2	62	10.4	3.2	117	9.0	3.2			
8	14.7	3.2	63	10.3	3.2	118	9.0	3.2			
9	14.4	3.2	64	10.3	3.2	119	9.0	3.2			
10	14.2	3.2	65	10.3	3.2	120	9.0	3.2			
11	14.0	3.2	66	10.2	3.2	121	9.0	3.2			
12	13.8	3.2	67	10.2	3.2	122	8.9	3.2			
13	13.7	3.2	68	10.2	3.2	123	8.9	3.2			
14	13.5	3.2	69	10.1	3.2	124	8.9	3.2			
15	13.4	3.2	70	10.1	3.2	125	8.9	3.2			
16	13.2	3.2	71	10.1	3.2	126	8.9	3.2			
17	13.1	3.2	72	10.1	3.2	127	8.9	3.2			
18	13.0	3.2	73	10.0	3.2	128	8.8	3.3			
19	12.9	3.2	74	10.0	3.2	129					
20	12.7	3.2	75	10.0	3.2	130					
21	12.6	3.2	76	9.9	3.2	131					
22	12.5	3.2	77	9.9	3.2	132					
23	12.5	3.2	78	9.9	3.2	133					
24	12.4	3.2	79	9.9	3.2	134					
25	12.3	3.2	80	9.8	3.2	135					
26	12.2	3.2	81	9.8	3.2	136					
27	12.1	3.2	82	9.8	3.2	137					
28	12.0	3.2	83	9.8	3.2	138					
29	12.0	3.2	84	9.7	3.2	139					
30	11.9	3.2	85	9.7	3.2	140					
31	11.8	3.2	86	9.7	3.2	141					
32	11.8	3.2	87	9.7	3.2	142					
33	11.7	3.2	88	9.6	3.2	143					
34	11.6	3.2	89	9.6	3.2	144					
35	11.6	3.2	90	9.6	3.2	145					
36	11.5	3.2	91	9.6	3.2	146					
37	11.5	3.2	92	9.5	3.2	147					
38	11.4	3.2	93	9.5	3.2	148					
39	11.3	3.2	94	9.5	3.2	149					
40	11.3	3.2	95	9.5	3.2	150					
41	11.2	3.2	96	9.4	3.2	151					
42	11.2	3.2	97	9.4	3.2	152					
43	11.1	3.2	98	9.4	3.2	153					
44	11.1	3.2	99	9.4	3.2	154					
45	11.0	3.2	100	9.4	3.2	155					
46	11.0	3.2	101	9.3	3.2	156					
47	10.9	3.2	102	9.3	3.2	157					
48	10.9	3.2	103	9.3	3.2	158					
49	10.9	3.2	104	9.3	3.2	159					
50	10.8	3.2	105	9.3	3.2	160					
51	10.8	3.2	106	9.2	3.2	161					
52	10.7	3.2	107	9.2	3.2	162					
53	10.7	3.2	108	9.2	3.2	163					
54	10.7	3.2	109	9.2	3.2	164					
55	10.6	3.2	110	9.2	3.2	165					
56	10.6	3.2	111	9.1	3.2	166					

APPENDIX B July flows a) upstream of all mainstem Whychus Creek diversions, and b) at Sisters City Park downstream of all major diversions on Whychus Creek.

a



b



Stream Connectivity in Whychus Creek

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Abstract

UDWC selected stream connectivity as an indicator of restoration effectiveness in Whychus Creek. Fish passage barriers are the primary feature affecting connectivity in the creek. Monitoring the number of river miles of habitat opened to resident and anadromous fish through barrier removal and retrofit projects provides a measure of stream connectivity. UDWC surveyed fish passage barriers along the creek in 2009 prior to any barrier removals. We compared survey data to criteria established by both ODFW and NMFS to determine if inventoried barriers were passage barriers for anadromous and resident fish. A total of six barriers were found to limit connectivity in Whychus Creek, effectively dividing the creek into seven reaches of varying length from less than one mile to 15.5 miles. From 2009 to 2014 UDWC retrofitted or removed four passage barriers, including the last concrete dam on the creek, increasing miles accessible from the mouth of the creek from X in 2009 to 26.8 in 2014. One inventoried barrier was found not to present a barrier to fish passage. No new passage projects were completed in 2015. UDWC continues to actively engage water rights holders to provide passage at the final passage barrier by 2017. Removal of this barrier will provide access to 11.5 additional miles of habitat, restoring connectivity along the entire length of Whychus Creek historically accessible to resident and anadromous species.

Introduction

The extent of stream connectivity, as influenced by the existence, condition and location of fish passage barriers, was selected as an indicator to be tracked over time on Whychus Creek. Although stream connectivity can be influenced by poor water quality or other habitat conditions, fish passage barriers are the primary feature affecting connectivity in Whychus Creek.

Fish passage barriers are widely recognized as hindering habitat connectivity by obstructing movement of aquatic species with the presence of physical barriers, changing velocities, water quality conditions and overall hydraulic and thermal alterations (Bergkamp *et al* 2000). With this recognition comes the realization that habitat connectivity along river systems is essential to healthy ecological function (Cote *et al* 2009, Wiens 2002). Passage barriers are therefore a simple and effective indicator of determining how much habitat is available to resident and anadromous fish species in Whychus Creek (Cote *et al* 2009). UDWC and its partners are working with landowners and water right holders to provide unimpeded up- and downstream fish passage by retrofitting or removing all fish passage barriers in Whychus Creek by 2017.

Monitoring the river miles of habitat opened to resident and anadromous fish through barrier removal provides a measure of stream habitat connectivity. Fish population data may become available to indicate whether anadromous and resident fish are accessing that habitat. While physical barriers such as dams limit accessibility to fish habitat, stream conditions including habitat quality and water quality can also function as passage barriers in limiting access to upstream and downstream habitat. Using fish passage barriers, fish population data, and habitat quality as indicators will help determine whether physical barriers alone are limiting movement of fish along Whychus Creek. The additional accessible river miles serve as a simple metric that allows effective communication of stream conditions to restoration partners and the general community.

UDWC used OWRD data and collected new data to establish a baseline number of barriers in Whychus Creek not meeting state and federal fish passage criteria as of 2009. We calculated the number of fragmented reaches created by the barriers and the number of stream miles for each reach. This report presents the status of fish passage and stream connectivity at the close of 2015.

Methods

The Oregon Water Resources Department (OWRD) inventoried water rights and associated diversion structures along the entire 40-mile length of Whychus Creek in 2002. Included in this inventory was information on location, presence of dams, pumps, headgates, fish screens and diversion size. Throughout 2008 and 2009, the locations of existing diversions identified in the OWRD 2002 survey were verified by field surveys. During this verification effort, sections of Whychus Creek between known diversion locations were surveyed to determine if any additional passage barriers existed.

Fish passage criteria are established by ODFW (ODFW) and are described in Oregon Administrative Rules (OAR) 635, Division 412 (ODFW 2009). In addition, NMFS has established fish passage criteria for anadromous species (2008). Many of the passage barrier structures in Whychus Creek are seasonal in nature and are often constructed of native materials available on hand. Push up dams constructed of river gravels and sediment are good examples of seasonal-type passage barriers. In addition, due to the high variability of flow conditions in Whychus Creek on a seasonal and diurnal level, hydraulic conditions vary greatly. In many instances, structures may meet fish passage criteria under certain flow conditions and seasons and not at others. As a result of these conditions, the inventoried fish passage barriers were classified as either meeting or not meeting ODFW and NMFS criteria for fish passage at the time of the baseline assessment. Barriers were defined as partial barriers if they were determined to allow fish passage at some times of year or for some life stages.

Data Collection

Baseline data were collected between 2002 and 2009 by OWRD and UDWC staff. Key information included latitude and longitude, river mile, date of survey, and barrier height along with pertinent comments relating to the barrier. Data were collected using a handheld GPS device along with measuring tapes and staffs for barrier configuration data. Water right holders were also interviewed to determine how diversions and barriers are operated throughout the year. This information was helpful in determining if barriers were passable for anadromous and resident species at any time throughout the year.

Data Analysis

Baseline data were compared to criteria established by both ODFW and NMFS (ODFW 2004, NMFS 2008) to determine if inventoried barriers were indeed passage barriers for anadromous and resident fish. Key criteria and parameters needed to satisfy fish passage include:

(1) Water velocity going over the barrier: must be ≤ 4 ft/sec (adults) and ≤ 2 ft/sec (juveniles)
(2) Channel water depth upstream of barrier: must be ≥ 8 inches

(3) Channel water depth downstream of barrier: must be ≥ 24 inches
(4) Water elevation difference above and below hydraulic jump: must be ≤ 6 inches

Criteria (3) and (4) are the main criteria that established whether barriers blocked anadromous and resident fish passage. It is important to note that not all barriers present fish passage barriers at all times of the year. Based on flow conditions and barrier operation (i.e. irrigation diversion dams), instances occur where passage at barriers is provided at different times of year. A barrier was considered a fish passage barrier if it did not meet the above ODFW and NMFS criteria at any time of the year.

New fish passage projects are designed and constructed to meet ODFW and NMFS criteria. UDWC inventoried fish passage projects completed from 2009 to 2015 and tallied projects under active development at the end of each calendar year. A project was defined as under active development if conversations with landowners and water rights holders had been initiated and were ongoing.

Results

The initial inventory completed in 2009 identified six fish passage barriers along Whychus Creek from river mile 15.5 to river mile 26.8 (Table 1, Figure 1). Barriers No. 1 and 2 were partial barriers, allowing anadromous fish at least intermittent access to a total of 23.8 miles of habitat. In December 2010 the Three Sisters Irrigation District dam fish passage restoration project was completed to meet fish passage criteria. Restoration of fish passage at the TSID dam connected adjacent 1.4- and 1.6-mile sections to create a three-mile reach and reduced the total number of reaches to six (Figure 2).

During OWRD and UDWC baseline inventories of existing fish passage barriers, surveyors had been unable to establish communication with the landowner and holder of a known diversion right. Because the water right was known to exist, an irrigation diversion and a barrier to fish passage was conservatively presumed to also exist. A detailed 2011 phone conversation with the landowner indicated that the barrier in question (Barrier No. 1) did not pose a barrier to fish passage. Full removal of the barrier was confirmed by the OWRD Basin Watermaster (J. Giffin, personal communication, July 2014).

From 2009 to 2015 fish passage was restored (or found to exist) at five of the six passage barriers, reducing the number of fragmented reaches to two and providing access to 26.8 miles from the mouth of the creek to the last artificial passage barrier on the creek, the McCallister irrigation diversion dam (Barrier No. 6). In 2015, UDWC staff performed an initial assessment of the diversion and irrigation water use at the last remaining diversion on Whychus Creek, barrier No. 6, and made contact with the landowner. UDWC will continue to advance development of a plan for the future of the diversion and irrigation water use to provide passage and access to the final 11.5 miles below the natural barrier of Whychus Falls.

Table 1.
Passage barrier specifications and status as of 2015. UDWC collected baseline data on passage barriers in Whychus Creek in 2009. Data that were not available were estimated based on OWRD surveys completed in 2002.

Barrier ID	Baseline Sampling Date	River Mile	Lat	Lon	Span (% of creek)	Dam height (ft)	Jump Height (inches)[1]	Jump Pool Depth (inches)[2]	Baseline Passage Barrier (Yes/No)	Notes	Passage Restored (Date)
No. 1	9/30/2002	15.5	44.3292	-121.4930	100%	2.0	No Data	No Data	Yes	Meyer push up diversion dam made of native materials. Passage Barrier determination established by OWRD	Dec-11
No. 2	8/28/2009	22.2	44.2858	-121.5485	100%	5.0	72.0	12.0	Yes	Leithauser Diversion Dam. Passage provided from April-Oct 15. Passage not provided Oct 15 - April across heavily degraded dam spillway.	Oct-13
No. 3	4/3/2009	22.6	44.282	-121.5531	100%	2.5	36.0	18.0	Yes	Sokol dam once used to create a backwater for fish rearing. No longer used and not associated with an irrigation water right.	Oct-11
No. 4	4/3/2009	23.8	44.2678	-121.5584	100%	4.5	48.0	18.0	Yes	Sokol irrigation diversion dam.	Sep-14
No. 5	8/28/2009	25.2	44.2515	-121.5502	100%	N/A	≤ 6.0	N/A	Yes	Three Sisters Irrigation District Dam. Channel raised to dam height and riffle created. Tallest feature height ≤ 6.0.	Dec-10
No. 6	8/28/2009	26.8	44.2356	-121.5633	100%	3.2	45.0	43.0	Yes	McCallister irrigation diversion dam	

[1] Water elevation difference above and below the hydraulic jump. **Must be ≤ 6 inches**

[2] Depth of water in plunge pool downstream of hydraulic jump. **Must be ≥ 24 inches**

Reference: *NMFS (National Marine Fisheries Service). 2008. Anadromous Salmonid Passage Facility Design. NMFS, Northwest Region, Portland, Oregon.*
ODFW (Oregon Department of Fish and Wildlife). 2004. Fish Passage Barrier Criteria

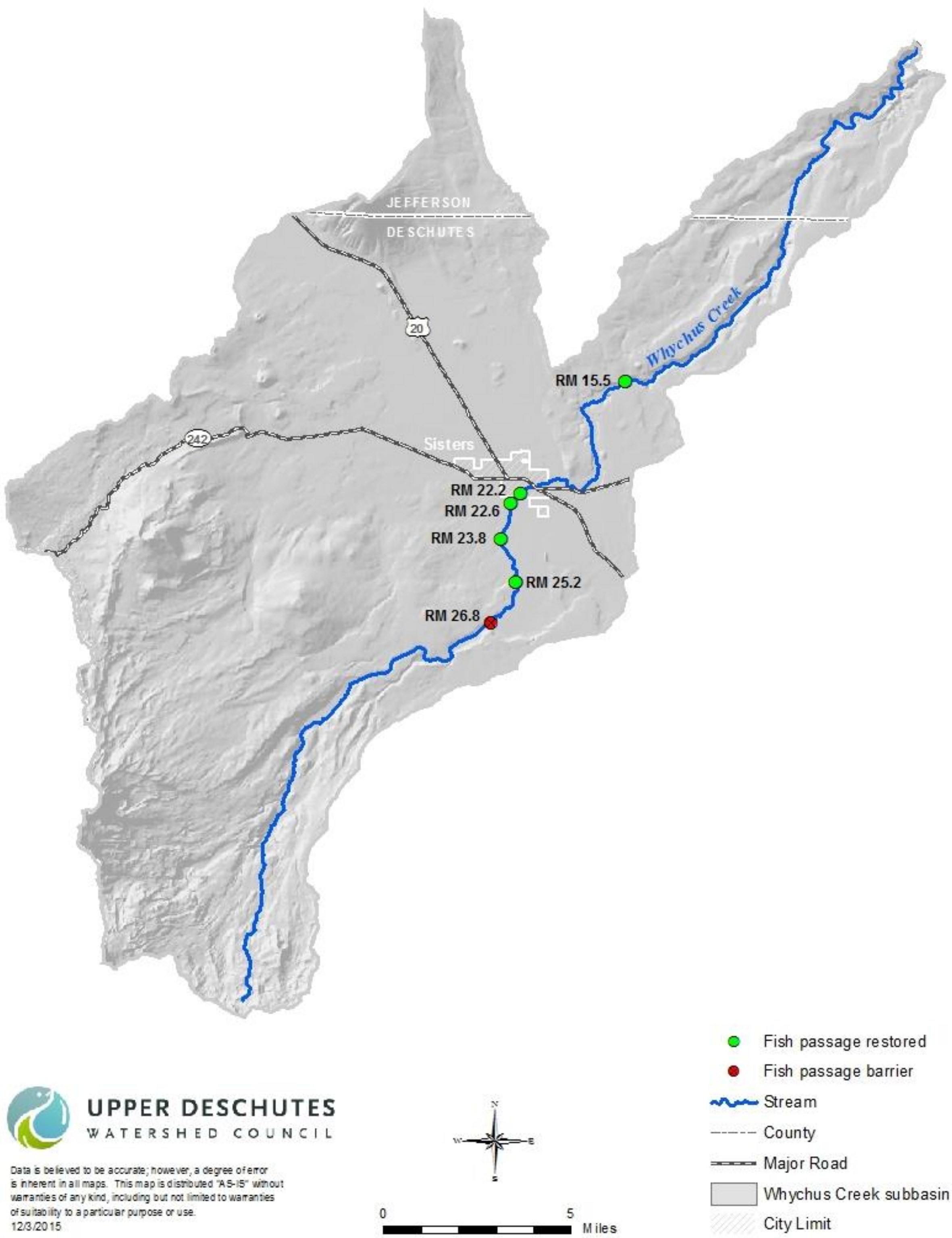

Figure 1.

In 2009, six barriers impaired stream connectivity on Whychus Creek between river miles 15.5 and 26.8. Over five years, from 2009 to 2015, fish passage was restored or found to be intact at five of these barriers. UDWC and restoration partners aim to provide passage at the one remaining barrier by 2017.

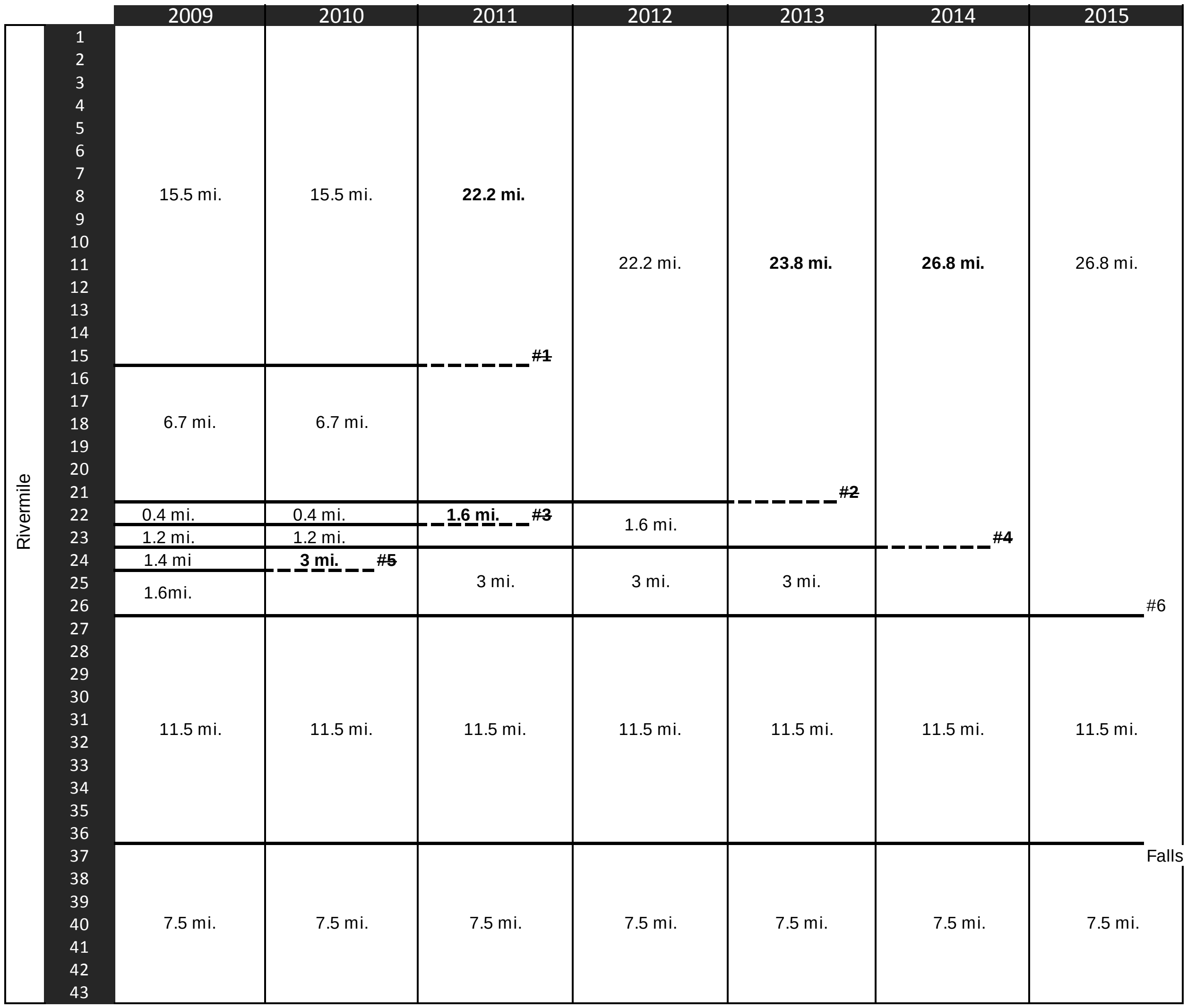

Figure 2.

Only one barrier to fish passage remains as of 2015, reducing the number of fragmented reaches from a baseline of seven to two: 26.8 miles accessible from the mouth of the creek, and 11.5 miles above the last remaining barrier. Whychus Creek Falls, located between river miles 36 and 37, is the downstream-most natural barrier to fish passage.

Discussion

Existing barriers determine the number of miles of contiguous stream habitat accessible to fish. Habitat connectivity has increased as barriers have been removed. UDWC continues to actively engage with the water rights holder at the last remaining barrier to develop and implement a project that will restore fish passage by 2017. Removal of this barrier will provide access along the entire length of the stream up to the natural barrier to fish passage at Whychus Falls, restoring connectivity along the entire length of stream habitat historically accessible to resident and anadromous species.

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Fish Entrainment Potential in Whychus Creek

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Abstract

The Upper Deschutes Watershed Council (UDWC) selected fish entrainment potential as an indicator of restoration effectiveness on Whychus Creek. UDWC described fish entrainment potential by the presence and location of irrigation diversions lacking state and federally approved fish screens. In the absence of detailed knowledge of the impacts of each diversion, the UDWC selected two simple metrics to estimate entrainment potential. The number of unscreened diversions and the total diversion rate associated with the irrigation diversion serve as coarse but cost-effective indicators of entrainment potential. In 2009 UDWC completed a baseline inventory of irrigation diversions on Whychus Creek. This inventory identified 13 active irrigation diversions along Whychus Creek, of which 12 did not have state and federally approved fish screens. Of the roughly 193 cfs of water diverted for irrigation, only 0.45 cfs or 0.02% of the water diverted for irrigation was diverted through state and federally approved fish screens as of 2009. From 2009 to 2015, restoration partners screened or decommissioned five diversions and reduced the total diversion rate by 28.6 cfs through water rights transactions. Since 2013, one diversion has been inactive and a percentage of the water right has been leased at a second diversion, further reducing the total diversion rate by 1.04 cfs. Together these actions have reduced the cumulative unscreened diversion rate by 86% from 2009 to 2014, from 192.89 to 22.40 cfs. Potential for fish entrainment in irrigation diversions has thus been substantially reduced. UDWC and restoration partners will continue to engage with water rights holders and landowners to eliminate risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

Introduction

UDWC selected fish entrainment potential as an indicator of restoration effectiveness on Whychus Creek. UDWC described fish entrainment potential by the presence and location of irrigation diversions lacking state and federally approved fish screens. Irrigation diversions can create two types of problems for fish. First, they potentially block upstream and downstream fish passage. Second, unscreened diversions divert fish almost as effectively as they divert water. This technical report updates the potential for fish entrainment at irrigation diversions in Whychus Creek at the close of 2015.

Numerous studies have shown that unscreened irrigation diversions act as sinks for fish populations (Roberts and Rahel 2008, Gale *et al* 2008, Carlson and Rahel 2007). The number of fish entrained into irrigation diversions in Whychus Creek prior to implementation of screening projects is unknown. However, within the Three Sisters Irrigation District canal, one of the major irrigation diversions on Whychus Creek, more than 5,000 fish were rescued in 2006 (M. Riehle, personal communication, 2009).

Screening irrigation diversions with state and federally approved screens reduces the potential for fish entrainment. Gale *et al* (2008) found that fish screens reduced or eliminated fish entrainment in one heavily managed stream in Montana, Skalkaho Creek. They found inter- and intra-annual variations in the proportion of fish entering diversions, and they suggested that variations in the proportion of water diverted accounted for some of the inter-annual variations in the number of fish diverted.

The location, design, timing, and volume of an irrigation diversion may affect its potential to entrain fish. In the absence of detailed knowledge of the impacts of each diversion, UDWC selected two simple metrics to estimate entrainment potential. The number of unscreened diversions and the total diversion rate associated with each diversion serve as coarse but cost-effective indicators of entrainment potential. By reducing the amount of water diverted through unscreened diversions, UDWC will decrease the magnitude of one factor limiting fish populations.

Methods

The Oregon Water Resources Department (OWRD) inventoried water rights and associated diversion structures along the entire 40-mile length of Whychus Creek in 2002. This inventory included information on diversion location, presence of dams, pumps, headgates, fish screens and diversion size. This dataset provided the basis for data collection efforts related to fish entrainment. Throughout 2008 and 2009, the UDWC conducted field surveys to verify the locations of existing diversions identified in the 2002 OWRD survey.

Fish screening criteria for the State of Oregon are established by ODFW (ODFW) and NMFS (National Marine Fisheries Service). NMFS establishes fish screening criteria for anadromous species (NMFS 2008) and ODFW currently follows NMFS criteria. New fish screens are built to specifications that meet ODFW and NMFS fish screening criteria.

Data Collection

OWRD and UDWC staff collected baseline data for irrigation diversions and screens along Whychus Creek from 2002 through 2009. Key information included latitude and longitude, river mile, date of survey, type of diversion and fish screening status along with pertinent comments relating to the fish screen. Data were collected using a handheld GPS device along with measuring tapes used to measure screen configurations. Water right holders were also interviewed to determine how diversions and barriers are operated throughout the year. DRC staff inventoried OWRD water rights and calculated the associated diversion rate for all diversions along Whychus Creek.

Data Analysis

UDWC compared baseline diversion screening data to screening criteria established by both ODFW and NMFS (NMFS 2008). We determined whether inventoried irrigation diversions did indeed provide adequate fish screening for anadromous and resident fish. While some irrigation diversions did have fish screens, the screens themselves may not have been state and federally approved.

UDWC collected, summarized and analyzed this data. Irrigation diversions were classified as either meeting or not meeting state and federal fish screening criteria for both anadromous and native resident fish species. As one method of establishing a baseline for risk of fish entrainment, the flow rate associated with each diversion as well as the total flow rate of unscreened water was tallied. As UDWC and restoration partners implement screening projects to meet state and federal critera, the total flow rate of unscreened irrigation water diverted from Whychus Creek will decline, signaling an associated reduction in the potential for fish entrainment.

Results

The 2009 baseline inventory identified 13 active irrigation diversions extending from river mile 9.8 to river mile 26.9, of which 12 did not have state and federally approved fish screens (Table 1, Figure 1). In 2009 the cumulative maximum irrigation diversion rate through unscreened diversions on Whychus Creek was 192.89 cfs representing almost 100% of the 193.34 cfs total diversion rate associated with existing water rights, and over 90% of the total streamflow during low flow periods in the summer and fall (Table 1). Of the roughly 193 cfs of water diverted for irrigation in 2009, only 0.45 cfs or 0.02% of the water diverted for irrigation was diverted through state and federally approved fish screens.

From 2009 to 2015, the total diversion rate was reduced by 28.6 cfs, from 193.3 cfs to 164.7 cfs, through streamflow restoration achieved by Deschutes River Conservancy, Three Sisters Irrigation District, and individual water rights holders (Table 1). Five diversions, including the TSID diversion representing the single largest flow withdrawal on Whychus Creek, were screened or decommissioned, reducing flows diverted through unscreened diversions by 86% (141.3 cfs), to 22.5 cfs. One abandoned diversion (Diversion No. 1, Plainview) was filled by USFS in 2012 to construct a trail, incidentally eliminating risk of fish entrainment at the former diversion. The presence of a screen in good condition at Diversion No. 11, although unknown to meet screening criteria, was confirmed by the OWRD Basin Watermaster (J. Giffin, personal communication, July 2014), suggesting the diversion likely poses a minimal risk for fish entrainment. Given the very low risk of fish entrainment posed by the presence of a screen in good condition and an associated diversion rate of 0.05 cfs, we report this diversion as screened despite the uncertainty about whether the screen technically meets state and federal screening criteria.

The Whychus Canyon restoration project will screen or decommission points of diversion No.s 12 and 13, in project reaches 2 and 5, respectively, to eliminate the risk of fish entrainment at these sites, concurrently reducing the cumulative rate of diversion through unscreened diversions by 1.04 cfs. Reach 2 implementation is scheduled for 2018; Reach 5 implementation is anticipated to break ground in 2022.

UDWC and ODFW, along with many of their partners, continue to actively work with landowners and water right holders to reach agreements to screen the remaining irrigation diversions to meet state and federal criteria and reduce the risk of entrainment for both anadromous and native fish species. As of the end of 2015 conversations with water managers and water rights holders to address fish screening at Diversion No. 2 were ongoing. Screening or decommissioning the three diversions currently in various stages of planning will reduce the total number of unscreened diversions to two.

Table 1.

ODFW and the Upper Deschutes Watershed Council surveyed diversions along Whychus Creek to establish a baseline inventory. The Upper Deschutes Watershed Council identified which diversions met state and federal criteria for fish screens as a proxy for fish entrainment potential. Four of the original twelve unscreened diversions were screened, one diversion was decommissioned, and one was replaced with a pump, between 2009 and 2014, leaving six diversions unscreened.

Diversion ID	Baseline Sampling Date	River Mile	Diversion Type	2009 Associated Diversion Rate (cfs)	Screen Present at Baseline Inventory	Screen opening size (inches)	Met State & Federal Criteria at Baseline Inventory	2015 Associated Diversion Rate (cfs)	Screened to meet criteria (date)	Meets State & Federal Criteria	Notes
No. 1	8/28/2009	25.25	Gravity	3.88	No	N/A	No	3.88	September-12	Yes	Plainview. Decommissioned, water rights consolidated.
No. 2	8/28/2009	25.15	Gravity	21.59	No	N/A	No	21.59		No	McCallister
No. 3	8/28/2009	23.90	Gravity	5.52	No	N/A	No	0.00	May-12	Yes	Lazy Z / Uncle John
No. 4	8/28/2009	23.65	Gravity	153.00	No	N/A	No	132.00	Apr-11	Yes	TSID
No. 5	8/28/2009	23.65	Gravity	1.00	No	N/A	No	1.00	Oct-10	Yes	Edgington
No. 6	8/28/2009	22.30	Gravity	5.00	No	N/A	No	4.00	Apr-14	Yes	Sokol
No. 7	8/28/2009	20.90	Gravity	1.12	No	N/A	No	0.00	Oct-09	Yes	Leithauser
No. 8	8/28/2009	18.65	Pump	0.07	Yes	1/4	No	0.07		No	No. 9 on OWRD List
No. 9	8/28/2009	18.15	Pump	0.38	Yes	1/4	No	0.38		No	Bradley
No. 10	8/28/2009	17.50	Pump	0.45	Yes	3/32	Yes	0.45	Aug-09	Yes	Deggendorfer
No. 11	9/30/2002	14.75	Pump	0.05	Yes	No Data	No	0.05	**Jul-16**	Unknown	Meyer. Screened, but unknown whether screen meets criteria.
No. 12	9/24/2002	11.20	Gravity	0.68	No	N/A	No	0.68		No	Remund. Not diverted in 2014; will be removed or screened within Whychus Canyon project
No. 13	9/24/2002	9.25	Gravity	0.60	No	N/A	No	0.60		No	Baker.0.24 cfs leased instream in 2015; diversion will be removed or screened within Whychus Canyon project
			Baseline Diversion Total	**193.34**			**2015 Diversion Total**	**164.70**			
			Baseline Unscreened Total	**192.89**			**2015 Unscreened Total**	**22.40**			

Reference: *NMFS (National Marine Fisheries Service). 2008. Anadromous Salmonid Passage Facility Design. NMFS, Northwest Region, Portland, Oregon.*

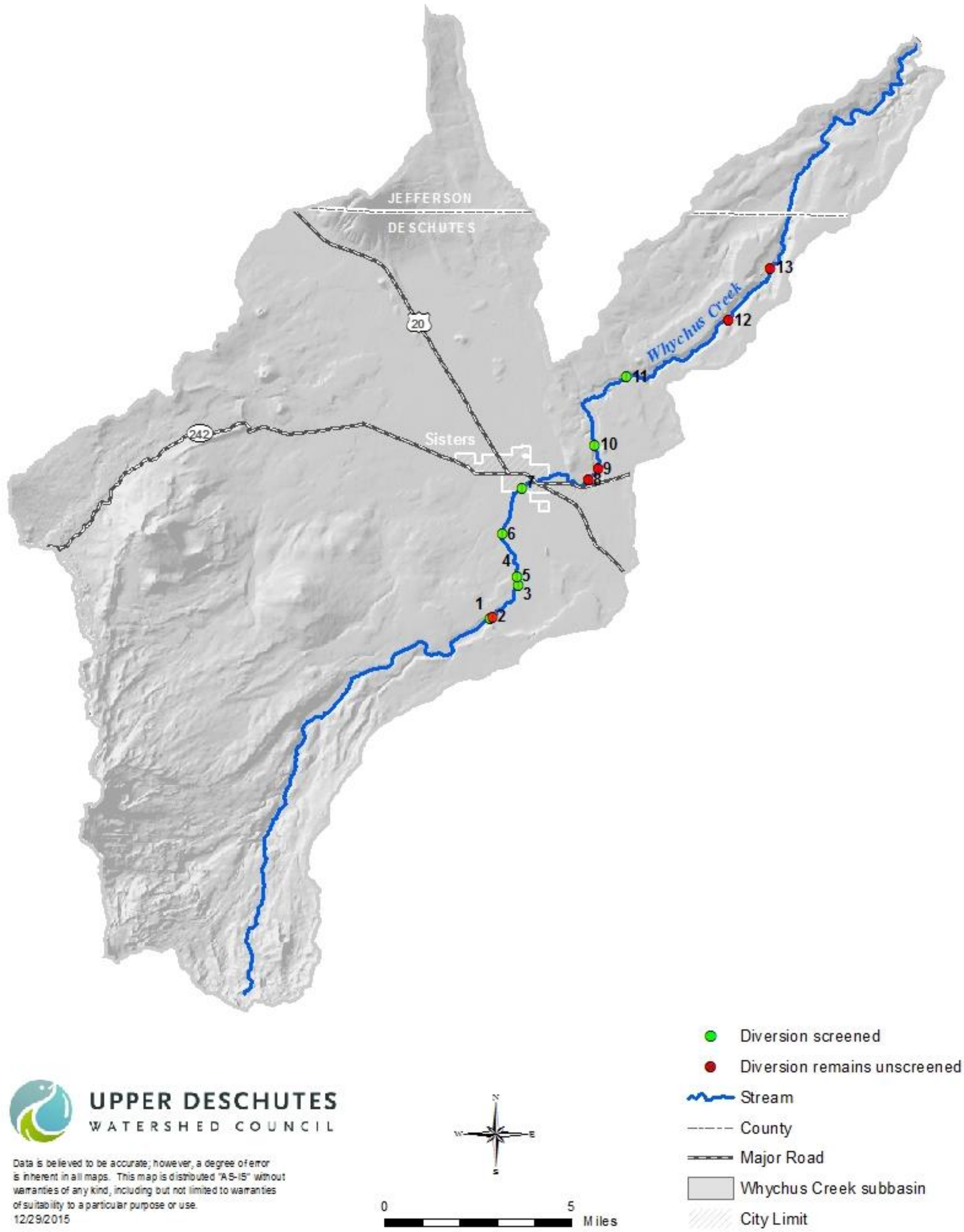

Figure 1.
The 2009 baseline inventory reported 12 of 13 irrigation diversions on Whychus Creek not to meet federal and state criteria for fish screens (NMFS 2008). From 2009 to 2015 cumulative flows diverted through unscreened diversions were reduced by 86%, to 22.40 cfs. Although the screen at Diversion no. 11 has not been verified to meet screening criteria, it is not considered to pose a risk of fish entrainment. Diversions No. 4 and 5 are located within feet of each other on opposite sides of the creek, thus only one of the two points is visible on the map.

Discussion

Although actual fish entrainment potential or risk associated with irrigation diversions takes into account a number of factors including diversion timing, location, structure, design, and geomorphology of the creek (i.e. pool, riffle, etc.), the cumulative diversion rate through unscreened diversions on Whychus Creek is one method to characterize fish entrainment risk to anadromous and native species. Because UDWC and restoration partners aim to implement projects to meet state and federal screening criteria at all irrigation diversions on Whychus Creek, and given that UDWC and its partners have not sought to prioritize which unscreened irrigation diversions pose the most significant fish entrainment risk, cumulative diversion rates through unscreened diversions represent a good metric for determining progress on reducing fish entrainment over time.

At the close of 2015, the cumulative unscreened diversion rate had been reduced by 86%, from 192.89 in 2009 to 22.40 cfs in 2015. Potential for fish entrainment in irrigation diversions has thus been dramatically reduced, owing greatly to the progressive practices of TSID management and to collaboration by water rights holders. UDWC and restoration partners will continue to engage with water rights holders and landowners to eliminate risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

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Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, and 2011-2015

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Abstract

This project uses aquatic macroinvertebrates as biomonitoring tools to assess the ecological effects of multiple restoration activities in Whychus Creek in Sisters, OR. Benthic macroinvertebrates were collected via kick net sampling from riffles in 10-13 reaches along Whychus Creek from RM 30.25 to RM 0.5 in late August of 2005, 2009, and 2011-2014. Xerces staff trained local volunteers in standardized sampling techniques for wadeable streams used by the Oregon Department of Environmental Quality (OR DEQ), and these trained teams collected macroinvertebrate samples from riffles at pre-determined sites. Samples from each reach were sub-sampled to a target count of 500 organisms and identified to the lowest practical level of taxonomic resolution (generally genus).

Analysis of the macroinvertebrate community in Whychus Creek using the PREDATOR predictive model and invertebrate-based indices of biotic integrity developed by OR DEQ, and a suite of taxonomic and ecological traits, reveal multiple changes since 2005. Similarity between duplicate samples taken by volunteers and Xerces staff at two sites in each year for quality assurance indicates that volunteers implement the protocol successfully, such that changes in the macroinvertebrate community are due more to altered habitat conditions than differences between operators. The community in 2015 appears to have been affected by the exceptionally dry hot spring and summer, as well as newly-implemented restoration work in the upstream reaches, since organismal abundance, community richness, and diversity of the sensitive EPT (Ephemeroptera, Plecoptera, and Trichoptera) were significantly lower among upstream reach sites compared to previous years. Even with these additional stressors, however, site IBI scores in 2015 were the same as or higher than the 2014 scores for seven of the 13 sampling sites, with 10 sites scoring as having slight disturbance, and three as having moderate disturbance. PREDATOR observed/expected (O/E) scores in 2015 were the same as or higher than the 2014 scores at 11 of the 13 sites, with three sites scoring as having poor biological condition, seven sites as fair, and three as good—the first “good biological condition” scores seen among any sites since 2009.

No dramatic changes in IBI or PREDATOR scores have occurred throughout this study. However, taxa identified by the PREDATOR model as “increasers” in each year include several genera of EPT and increased numbers of OR DEQ cool temperature indicators, while decreaser communities have contained increasing numbers of warm temperature and high sediment indicator taxa, and the mean percent fine sediment (%FSS) optima have been lower in the increaser community compared to the decreaser community in every year. Missing and replacement taxa identified by PREDATOR in each sampling year have included both sensitive and tolerant taxa, but a shift has occurred from earlier years in which replacement taxa had higher mean temperature optima than missing taxa, to a significantly lower temperature optima among replacement compared to missing taxa followed by no strong differences between the two. The %FSS optima of the replacement community have been significantly lower than that of the missing community in nearly every sampling year.

Overall, taxonomic differences are greatest between the 2005/2009 macroinvertebrate community and all subsequent sampling years, including 2015, with changes including a greater abundance of sensitive and/or lotic-associated taxa. This is further reflected by a sustained decrease in mean temperature optima for the communities at downstream and mid-stream sampling reaches across time, and an initial decrease in mean %FSS optima of the stream community. Analysis of functional feeding groups (FFGs) has been less informative, with more generalist feeding groups (predators and collector-gatherers) dominating the community in all years.

New taxa are collected in each year; most are in the sensitive EPT group, including all three of the new taxa found in 2015. The number of new taxa has decreased with each sampling year, suggesting that colonists from the existing pool in surrounding habitats have filled available niches under altered stream conditions and that the community may be stabilizing, although continuing restoration work and increasingly severe climate conditions are likely to impose additional stresses. Overall, since restoration began in 2005, the diversity of sensitive taxa in Whychus Creek has increased, and the community has shifted towards taxa with lower temperature and sediment optima.

Project Background and Summary

Bioassessment

Freshwater ecosystems are severely impacted by human activities, with 45% of US waters classified as endangered or impaired (US EPA, 2013). Stream restoration projects have intensified, but determining their ecological success requires baseline assessment and long-term post-project monitoring of biotic communities (biomonitoring), and an analysis of how changes in community composition relate to stream ecosystem functions (Bernhardt et al., 2005). Because biomonitoring evaluates the condition of biological communities inhabiting a water body, such as plants, amphibians, algae, diatoms, or invertebrates, it provides data about stream "health" that physical and chemical data alone do not address (Rosenberg & Resh, 1993; Karr & Chu, 1999). The structure of biological communities changes in response to habitat impairment, based on individual species' sensitivity or tolerance to different stressors. The communities assessed must generate a biological "signal" based on human impacts that can be detected apart from the "noise" of normal variation in space and time (i.e. changes in season or stream order). Benthic macroinvertebrates (BMIs) are useful for biomonitoring because they are critical to the food web, are confined to water for most or all of their life cycle, respond in a known way to a range of human-induced stressors, have a short generation time that allows changes in community structure to be detected rapidly, are ubiquitous and abundant, and their sampling and identification is relatively straightforward, standardized, and cost-effective.

Many restoration projects are undertaken with the assumption that improved physical habitat automatically increases faunal biodiversity, which in turn restores impaired or lost ecological processes, i.e. "the field of dreams hypothesis" (Palmer et al., 1997). However, a variety of reach- and catchment-specific influences must be considered when evaluating project outcomes (Roni et al., 2002; Bond & Lake, 2003; Palmer et al., 2005; Lake et al., 2007. Restoration can improve habitat and water quality at the reach level, but streams experience watershed-wide stressors that site-specific activities may not completely remediate (Booth & Jackson, 1997; Bohn & Kershner, 2002; Bond & Lake, 2003). The composition and mobility of the regional pool of colonists must also be considered when assessing post-restoration changes in biological communities (Bond & Lake, 2003; Blakely et al., 2006), and because the recovery process can take years, a long-term monitoring plan is imperative.

Whychus Creek Project

The aquatic macroinvertebrate community in Whychus Creek was sampled in 2005, 2009, and 2011-2015 at 10-13 reaches from RM 30.25 to RM 0.5, to assess changes in community composition following multiple restoration projects. This long-term effectiveness monitoring also engages local stakeholders, as annual macroinvertebrate sampling is done by community volunteers trained by Xerces staff. This study uses multiple approaches to examine the benthic macroinvertebrate community and the ecological implications of changes in community composition. Standard biotic assessment techniques, including the general invertebrate Index of Biotic Integrity (IBI), Grande Ronde IBI (GR-IBI), and the PREDATOR predictive model (Hubler, 2008), all developed by the OR DEQ, are applied annually to assign levels of biological impairment to sampling reaches. Changes in individual IBI metrics as well as community tolerances for percent fine sediment and temperature, relative diversity and abundance of OR DEQ temperature and sediment indicator taxa, and functional feeding group composition are also assessed. Multivariate analysis is done to identify taxa contributing the most to observed community differences from year to year.

Methods

Sampling Sites

Thirteen reaches were sampled along Whychus Creek in 2015, with duplicate samples taken at two sites for quality control (Table 1). Some sites have been added or removed from the slate since 2005 based on access, changes in land use, or re-assessment of their importance, and some site names changed following more precise GIS mapping done in 2014. Sampling sites are distributed broadly into downstream (RM 0.5- 11.5; DS), mid-stream (RM 18-19.5; MS), and upstream (RM 23.5 - 30.25; US).

Table 1. Whychus Creek sampling sites, 2005-2015.

Site ID	Description	Coordinates	Year(s) sampled
WC0050	RM 0.50	44.45682, -121.34028	2005
WC0150	RM 1.5, d/s Alder Springs	44.44491, -121.34543	2009, 2011-2015
WC0300[a]	RM 3, u/s Alder Springs	44.43458, -121.35976	2005
WC0600[b,d]	RM 6, u/s Rd 6360	44.40412, -121.40259	2005, 2009, 2011-2015
WC0650	RM 6.5, u/s Rd 6360 crossing	44.396799, -121. 404268	2013
WC0850[d,f]	RM 8.75, Rimrock Ranch d/s	44.391278, -121.406182	2011-2015
WC0900	RM 9, Rimrock Ranch	44.384198, -121.407892	2005, 2009, 2011-2015
WC0925 [e]	RM 9.25, Rimrock Ranch	44.380815, -121.408592	2013
WC1025	RM 10.25, Rimrock Ranch u/s	44.371534, -121.415865	2011-2012, 2014-2015
WC1100	RM 10.25	44.364587, -121.421706	2014-2015
WC1150	RM 10.75	44.361288, -121.427525	2014-2015
WC1800	RM 18	44.328342, -121.494534	2005
WC1825	RM 18.25, d/s end DBLT property	44.32781, -121.495406	2009, 2011-2015
WC1850	RM 18.5, DBLT property	44.326601, -121.500229	2009, 2011-2015
WC1900	RM 19, DBLT property	44.321523, -121.507461	2005, 2009, 2011-2015
WC1950[g]	RM 19.5, d/s Camp Polk Bridge, DBLT	44.318741, -121.514961	2009, 2011-2015
WC2350	RM 23.5, Perit Huntington Rd.	44.29066, -121.53064	2005
WC2425[c,e,g]	RM 24.25, City Park, d/s gauge	44.287806, -121.544229	2005, 2009, 2011-2015
WC2600[c,f]	RM 26, 4606 Rd. footbridge	44.2730592, -121.555297	2005, 2009, 2011-2015
WC2650	RM 26.5, d/s TSID	44.256434, -121.550692	2011
WC2700	RM 27, u/s TSID	44.250744, -121.549892	2011
WC3025	RM 30.25, OWRD gauge	44.233647, -121.567105	2005, 2009, 2011

Superscripts indicate sites where duplicate samples were taken for quality control: [a] 2005 duplicate; [b] 2009 duplicate; [c] 2011 duplicate; [d] 2012 duplicate; [e] 2013 duplicate [f] 2014 duplicate; [g] 2015 duplicate.

Volunteer Training

Volunteer training and stream sampling was done on a single day in the same index period across all years (August 17-20). Volunteers assembled at City Park (Sisters, OR) to be trained by Xerces staff in standardized macroinvertebrate monitoring protocols for Oregon's wadeable streams (OWEB, 2003). The sampling technique was demonstrated and each item on the data sheet was explained. A handout with detailed sampling instructions and field guides to Northwest stream macroinvertebrates (Adams *et al*., 2003) were provided, although volunteers did not identify organisms. The group was divided into teams of three to four people, each equipped with a D-frame kick net with 500 μm mesh; metal 500 μm sieve, 500 μm mesh Nitex membrane square; squirt bottle; plastic spoons; forceps; thermometer; tape measure; 10-gallon plastic bucket; hand lens; 1-liter Nalgene sample jars half-filled with 80% ethanol; datasheets; jar labels; clipboard; and detailed instructions for finding sites and the upstream and downstream ends of sampling reaches. Volunteers also had cell phone numbers of Xerces and UDWC staff, and staff moved among sites throughout the day to answer questions. Samples were returned to the park at the end of the day and inspected by Xerces staff to be sure each was properly labeled and preserved.

Sampling Techniques

Benthic macroinvertebrates were collected from riffle habitats according to standardized protocols developed by OR DEQ for Oregon's wadeable streams (see OWEB, 2003). Sample reach lengths were calculated as 40 times the average wetted width of the stream at the sampling point, with a minimum of

500 feet and a maximum of 1000 feet. Watershed council staff performed these calculations and flagged the upstream and downstream extent of each reach a few days prior to sampling.

A sample from a single reach consisted of composited net sets taken from eight different randomly selected riffles within the designated reach. Each of the eight net sets was collected from a 1 ft^2 area using a 500 μm D-frame kick net. In reaches with fewer than eight riffles, two kick net samples were taken in each of four riffles. Large rocks and debris in the sampling area were first rubbed and rinsed into the net to dislodge and collect any clinging organisms and set aside. The substrate was then disturbed thoroughly using a boot heel to a depth of 6-10 cm for 1-2 minutes. The eight net sets at each site were pooled into a bucket; large debris was rinsed and removed, and any vertebrates such as fish were noted and carefully replaced in the stream. Sample material was concentrated by being poured through a 500 μm sieve lined with a flexible square of 500 μm Nitex membrane; the membrane was then lifted out and the concentrated sample carefully scooped and rinsed into the sample jar.

Samples with excessive amounts of sand and/or gravel were elutriated to separate the lighter invertebrates from heavier mineral material and allow them to be jarred separately to avoid crushing or grinding specimens. Elutriation was done by adding water to the sample bucket, swirling it thoroughly, then pouring the suspended organic material through the sieve. After two to three rinses, the organic material was placed in one sample jar and the mineral material in another; all sample material from each site was retained for subsequent examination in the lab so that heavy-bodied organisms (i.e., snails, stonecase-making caddisflies) were not lost. Jars were filled no more than halfway with sample to ensure preservation, and the ethanol was replaced with within 48 hours to maintain the 80% concentration, since water leaches from the sample and dilutes the preservative. A simple physical habitat assessment was done at each site to provide data on human use and landscape alterations, substrate composition, water temperature and appearance, and wetted width and depth at each riffle sampled (see Appendix A for datasheet).

Macroinvertebrate Identification

Sample identification was done by Cole Ecological, Inc. Each composite sample was randomly sub-sampled to a target count of 500 organisms. If the sample contained fewer than 500 organisms, the entire sample was picked and identified. All organisms were identified to the level of taxonomic resolution currently used by the OR DEQ (genus and species). If a specimen was too immature for key taxonomic characters to be fully developed or visible, identification was done only to the family.

Data analysis

Invertebrate Index of Biotic Integrity (IBI)

Biological condition of each sampling site was assessed using both the OR DEQ general IBI and the more regional northeastern (Grande Ronde) IBI (Table 2), and the correlation between the two sets of scores was determined. Ranges used to calculate the scaled score of each metric in the IBIs were created based on community composition in reference streams determined previously by OR DEQ. A higher scaled score (5) is given to metric ranges typical of a healthy stream, while a lower scaled score (3 or 1) reflects values associated with more degraded conditions. Some metrics are thus positive (a higher raw value receives a higher scaled score) while others are negative (a higher raw value receives a lower scaled score). Scaled scores for all metrics are summed to generate a single value that reflects the level of site impairment.

All sites from all sampling years were assessed using both IBI models and changes across years in mean raw metric values as well as mean scaled values among upstream, mid-reach, and downstream sites. Two-tailed t-tests were done to determine significance of differences between years.

Table 2. OR DEQ general IBI and Grande Ronde IBI metrics and scoring. Both use taxa identified to genus.

	OR DEQ IBI				**Grande Ronde IBI**		
	Scoring Criteria						
Metric	5	3	1		5	3	1
POSITIVE METRICS							
Taxa richness	>35	19-35	<19		>31	24-31	<24
Mayfly richness	>8	4-8	<4		>7	6-7	<6
Stonefly richness	>5	3-5	<3		>6	5-6	<5
Caddisfly richness	>8	4-8	<4		>4	2-4	<2
# sensitive taxa	>4	2-4	<2		>4	3-4	<3
# sediment-sensitive taxa	≥2	1	0		>1	1	0
NEGATIVE METRICS							
% dominance*	<20	20-40	>40		<39	39-42	>42
% tolerant taxa	<15	15-45	>45		<24	24-36	>36
% sediment-tolerant taxa	<10	10-25	>25		<10	10-15	>15
MHBI**	<4	4-5	>5		<3.9	3.9-4.3	>4.3
Summed Score & Condition							
Severely impaired	<20				<15		
Moderately impaired	20-29				15-25		
Slightly impaired	30-39				N/A		
Minimally/not impaired	>39				≥26		

**For general IBI, dominance of the top (most abundant) taxon is assessed; for Grande Ronde IBI, abundance of the top 3 taxa is assessed.*

***MHBI = Modified Hilsenhoff Biotic Index, reflecting tolerance to organic pollution/enrichment; individual taxa MHBI values may range from 1 (low tolerance) to 10 (high tolerance).*

PREDATOR model

The PREDATOR model (Predictive Assessment Tool for Oregon; Hubler, 2008) was developed for two major regions in Oregon: Marine Western Coastal Forest (Willamette Valley and Coast Range ecoregions; MWCF) and Western Cordillera and Columbia Plateau (Klamath Mountain, Cascades, East Cascades, Blue Mountains, and Columbia Plateau ecoregions; WCCP). The model calculates the ratio of taxa observed at a sampling site to taxa expected at that site if no impairment existed (O/E), based on community data collected previously at a large number of reference streams. The model incorporates environmental gradients such as elevation, slope, and longitude to select the most appropriate reference streams. An O/E value of less than one indicates loss of taxa, while values greater than one indicate taxa enrichment, potentially in response to pollution or nutrient loading.

PREDATOR scores are generated from a habitat file and a macroinvertebrate data file for each site, which are uploaded to the model software at the Western Center for Monitoring & Assessment of Freshwater Ecosystems (www.qcnr.usu.edu/wmc). Model outputs include a site test result, which indicates whether the habitat data falls within the model parameters; an O/E score for each sample, which provides a measure of biological condition; a probability matrix indicating taxa expected to occur at each site but absent, and taxa not expected to occur at the site but which are present (missing and replacement taxa, respectively); and a sensitivity index that reveals "increaser" and "decreaser" taxa in the community. Invertebrate community data were submitted to the PREDATOR WCCP model (Hubler, 2008). O/E scores associated with a probability of capture (P_c) > 0.5 were used (i.e. the model considers only invertebrates with over 50% likelihood of being collected at reference sites). In the WCCP model, site biological condition is assigned based on the following O/E scores: $\leq$0.78 = poor (most disturbed); 0.79 – 0.92 = fair (moderately disturbed); 0.93 – 1.23 = good (least disturbed); and >1.23 = enriched.

Predictive models are often considered to be more sensitive and accurate than IBIs, but it should be noted that the PREDATOR model has not been re-calibrated since it was created using stream survey data from 1998-2004. Periodic sampling of reference streams used to build the models and model re-calibration is needed to detect any shifts arising from altered environmental conditions such as climate change, so it is possible that the sensitivity of this model has changed over time, especially as the WCCP model applicable to the Cascades has lower precision than the MWCF model developed for the Willamette Valley (Hubler, 2008; Hubler, pers. comm.). In addition, mean annual rainfall at the sites along Whychus Creek is at the very low end of the annual rainfall range at the reference sites from which community data are drawn for analysis (Hubler, pers. comm.), which may affect model accuracy.

Temperature and Sediment Optima

Differences in macroinvertebrate community composition may be driven by stream temperature and the amount of fine sediment in the substrate. OR DEQ developed a dataset of individual taxa optima values for seasonal maximum temperature and percent fine sediments (i.e. the temperature or %FSS under which a taxon can maximize its abundance). Temperature and sediment optima of increaser vs. decreaser taxa and missing vs. replacement taxa were examined to diagnose whether the community was responding to changes in temperature and/or sediment conditions. The weighted mean temperature and sediment optima of the macroinvertebrate community at each sampling site were also examined across sites and years. The presence of taxa considered by OR DEQ to be indicators (i.e. taxa with the strongest responses to environmental gradients) of cool or warm temperatures and high or low fine sediment conditions (Hubler et al., 2008; see Appendix B for a list of OR DEQ indicator taxa) was also noted among missing, replacement, increaser, and decreaser groups.

Taxonomic and Ecological Trait Analysis

Multimetric and multivariate models routinely examine taxonomic differences among biotic communities; the identity of species within the community serves as a surrogate for the attributes they possess that are affected by changing environmental conditions (Southwood, 1977). A prime illustration is the ubiquity of the “EPT” metric, which looks at the richness and/or abundance of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera), taxa considered as a whole to be the most sensitive to decreased flows and/or increased temperature, sedimentation, and pollution. However, ecological traits, which involve properties such as trophic guild, body size, or number of generations per year, can also be examined to assess site conditions in conjunction with taxonomic traits (Pollard and Yuan, 2010; Culp et al., 2011; van den Brink et al., 2011; Lange et al., 2014). Ecological traits provide additional insights into mechanisms structuring the community at a site, and assessing a combination of taxonomy- and biology-based traits can increase the likelihood of identifying recovery of macroinvertebrate assemblages (Arce et al., 2014).

To incorporate ecological traits into bioassessment of Whychus Creek, a functional feeding group (FFG) designation was assigned to each taxon in the dataset from 2005-2015. Designation of a taxon as a predator, scraper, shredder, collector-filterer, or collector-gatherer was done according to Merritt et al. (2008). Richness, relative diversity, and relative abundances of taxa in different FFGs was determined for each year’s dataset, and changes in distributions among downstream, mid-reach, and upstream sites were examined.

In addition, as the OR DEQ IBI incorporates both taxonomic and ecological (sensitivity/tolerance) metrics, changes across years in the raw scores of individual IBI metrics were examined for each site, as well as changes in the mean richness, relative diversity, and relative abundance of combined EPT taxa.

Community Similarity Analysis

Additional analyses to detect patterns in macroinvertebrate community composition were done using the PRIMER v6 ecological community statistics software package (Clarke & Warwick, 2001). CLUSTER analysis of a Bray-Curtis similarity matrix of square-root transformed abundance data was done to investigate macroinvertebrate community similarity between sites and across years. To examine the communities of increaser/decreaser and missing/replacement taxa, CLUSTER analysis was done on a presence/absence abundance dataset. Community similarity was also subjected to ordination by non-metric multidimensional scaling (MDS) to investigate site assemblage similarity based on sampling year and stream reach location. SIMPER analysis was used to assess the taxa that contributed the most to community differences between years.

Results and Discussion

Indices of Biotic Integrity

OR DEQ General IBI

In 2015, 10 of the 13 sites sampled (77%) received IBI scores indicating slight biological impairment, and three sites (23%) received scores indicating moderate biological impairment. These scores follow the trend seen in previous years, with most sampling sites scoring as slightly impaired, and the number of slightly impaired sites increasing over time while the proportion of moderately impaired sites has shown a slight downward trend (Figure 1). No sites have ever received a score indicating severe levels of impairment, but few to no sites can be considered unimpaired or minimally impaired under this IBI. The proportion of minimally impaired sites has ranged from 8-18% in past years, but since 2013, no sampling site has received a general IBI score high enough to qualify as minimally impaired (Table 3).

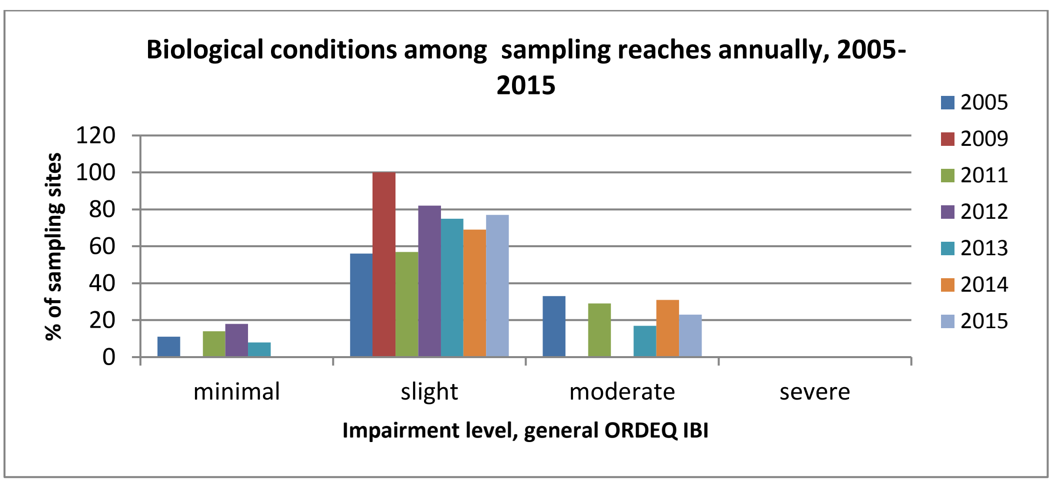


Figure 1. Comparison of site biological condition categories across time

Table 3. OR DEQ IBI scores for all sites sampled between 2005 and 2015. Colors indicate minimal (orange), slight (green), or moderate (blue) disturbance.

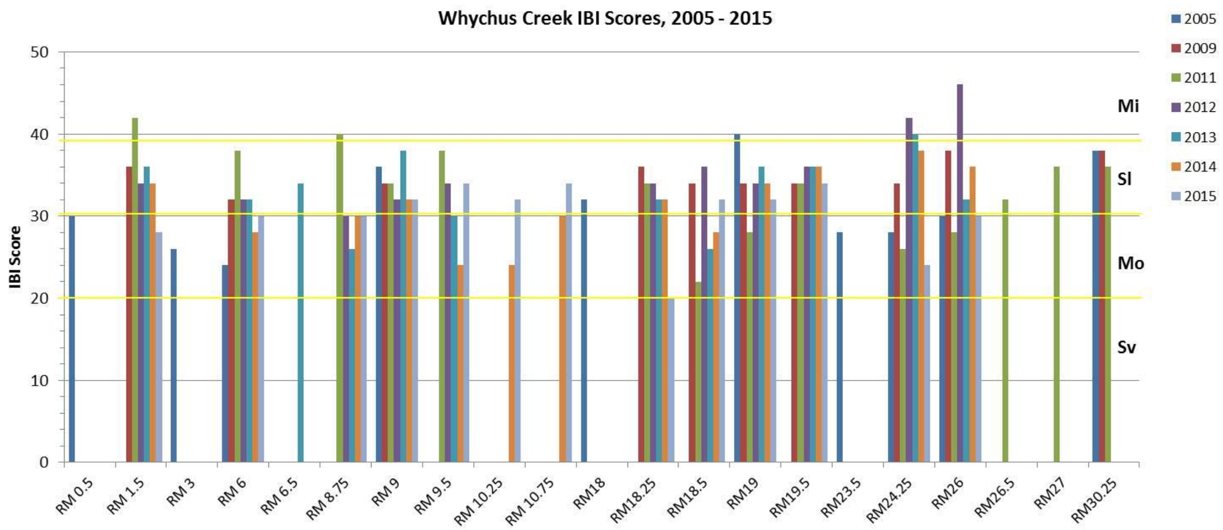
Site	2005	2009	2011	2012	2013	2014	2015
WC0050	30	---	---	---	---	---	---
WC0150	---	38	44	34	36	34	28
WC0300	26	---	---	---	---	---	---
WC0600	24	32	38	32	32	28	30
WC0650	---	---	---	---	34	---	---
WC0875	---	---	40	30	26	30	30
WC0900	36	34	34	32	38	32	32
WC0950*	---	---	38	34	30	24	34
WC1025 (WC1100)**	---	---	---	---	---	24	32
WC1075 (WC1150) **	---	---	---	---	---	30	34
WC1800	32	---	---	---	---	---	---
WC1825	---	36	34	34	32	32	20
WC1850	---	34	22	36	26	28	32
WC1900	40	34	28	34	36	34	32
WC1950	---	34	34	36	36	36	34
WC2325	28	---	---	---	---	---	
WC2425	28	34	26	42	40	38	24
WC2600	30	38	28	46	32	36	30
WC2650	---	---	32	---	---	---	---
WC2700	---	---	36	---	---	---	---
WC3025	38	38	36	---	---	---	---

*in 2012, sampling was done at RM 9.25 instead of RM 9.5
** Site names changed for these locations based on improved GPS mapping

Individual site scores have changed across time, but no sustained trend towards increased IBI scores has been seen for individual sites (Figure 2A). When considered across different stream reaches, IBI scores have been relatively stable across time among the downstream and midstream reach sites, with greater annual differences occurring at upstream sites (Figure 2B). Among downstream sites, mean IBI scores increased from 2005 to 2011, with the mean score in 2011 significantly greater than in 2005, and the impairment level also improved from moderate to slight. The mean IBI score among downstream reaches in every subsequent year has been significantly lower than in 2011, but mean scores within the 2012-2015 span have not differed significantly for the sites, and in all years except 2014, the mean score indicated only slight impairment among downstream sampling reaches. Similarly, although there has been some annual change in mean IBI score in the midstream reach, these differences have never been significant, and in three of the seven sampling years, the mean score IBI score has reflected slightly impaired conditions among midstream sites.

Among the upstream sampling reaches, annual differences in mean IBI scores have been greater. From 2005 to 2012, mean scores increased, and site conditions showed a significant improvement from slight to minimal impairment. Mean IBI scores were lower in both 2013 and 2014; though the difference was not significant compared to the mean score in 2012, but site conditions still registered as slightly impaired. In 2015, the mean IBI score among upstream reaches was significantly lower than in 2012 and 2009, which was reflected in the corresponding moderately impaired condition category. However, two factors may be influencing this lower mean score in 2015: 1. very low precipitation and an unusually warm winter and hot summer may have influenced flow and temperature conditions in the stream, affecting the macroinvertebrate community; and 2. one of the two upstream sampling reaches, WC2600, is the site of the Whychus Floodplain Project, and recently experienced extreme perturbations such as earth moving, flow interruption, and disturbance of the sediment regime.

a. Individual site scores across time. Mi = minimal impairment, Sl = slight, Mo = moderate, Sv = severe



b. Mean IBI scores across time among different stream reaches. Letter pairs indicate significant difference between means (p<0.05). Downstream = RM 0.5-11.5; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

Figure 2. OR DEQ general IBI scores for Whychus Creek sampling sites.

<u>Grande Ronde IBI</u>

Macroinvertebrate community data were also analyzed using a more regional IBI developed by OR DEQ for northeastern Oregon (Grande Ronde IBI; Shannon Hubler, pers. comm.). The metrics in the Grande Ronde (GR) and general IBI are the same, except that the general IBI scores the percent dominance of the single most abundant taxon while the GR IBI scores dominance of the three most abundant taxa. Raw score ranges and corresponding scaled scores differ between the two IBIs, and the GR IBI scores reflect only three levels of biological condition (minimal impairment, >25; moderate impairment, 15-25;

and severe impairment, <15), in contrast to the four categories in the general IBI (minimal, slight, moderate, and severe impairment; see Table 2).

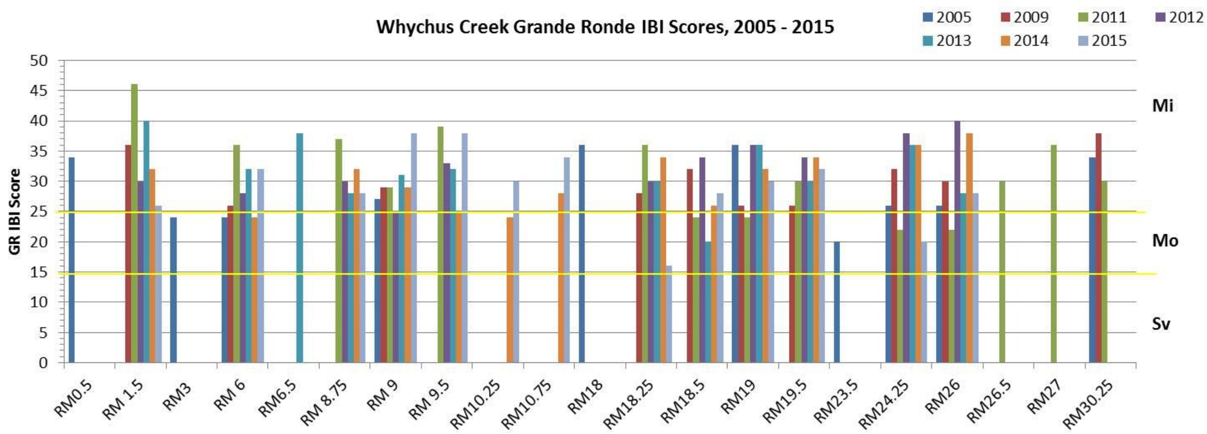
Most sites (70-100%) in all years scored as minimally impaired (Table 5; Figure 3A); the remainder scored as moderately impaired, and no site had a summed score low enough to indicate severe impairment. Annual changes in mean IBI score among the reaches varied when using the GR IBI (Figure 3B), but differences were generally not significant from year to year, and usually did not change the biological condition category. Although the GR IBI score tended to indicate better biological conditions, the scores for the general and GR IBI have a strong positive correlation ($r^2 = 0.6523$)

Table 5. Grande Ronde IBI scores for all sites sampled between 2005 and 2014. Colors indicate minimal (orange) or moderate (green) disturbance; no site scored as severely disturbed in any year.

Site	2005	2009	2011	2012	2013	2014	2015
WC0050	34	---	---	---	---	---	
WC0150	---	36	46	30	40	32	26
WC0300	24	---	---	---	---	---	---
WC0600	24	26	36	28	32	24	32
WC0650	---	---	---	---	38	---	---
WC0875	---	---	37	30	28	32	28
WC0900	27	29	29	25	31	29	38
WC0950*	---	---	39	33	32	25	38
WC1025 (WC1100)	---	---	---	---	---	24	30
WC1075 (WC1150)	---	---	---	---	---	28	34
WC1800	36	---	---	---	---	---	---
WC1825	---	28	36	30	30	34	16
WC1850	---	32	24	34	20	26	28
WC1900	36	26	24	36	36	32	30
WC1950	---	26	30	34	30	34	32
WC2325	20	---	---	---	---	---	---
WC2425	26	32	22	38	36	36	20
WC2600	26	30	22	40	28	38	28
WC2650	---	---	30	---	---	---	---
WC2700	---	---	36	---	---	---	---
WC3025	34	38	30	---	---	---	---

*in 2012, sampling was done at RM 9.25 instead of RM 9.5

a. Individual GR IBI site scores across time



b. Mean GR IBI scores among different stream reaches. Letter pairs indicate significant difference between means (p<0.05). Downstream = RM 0.5-11.5; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

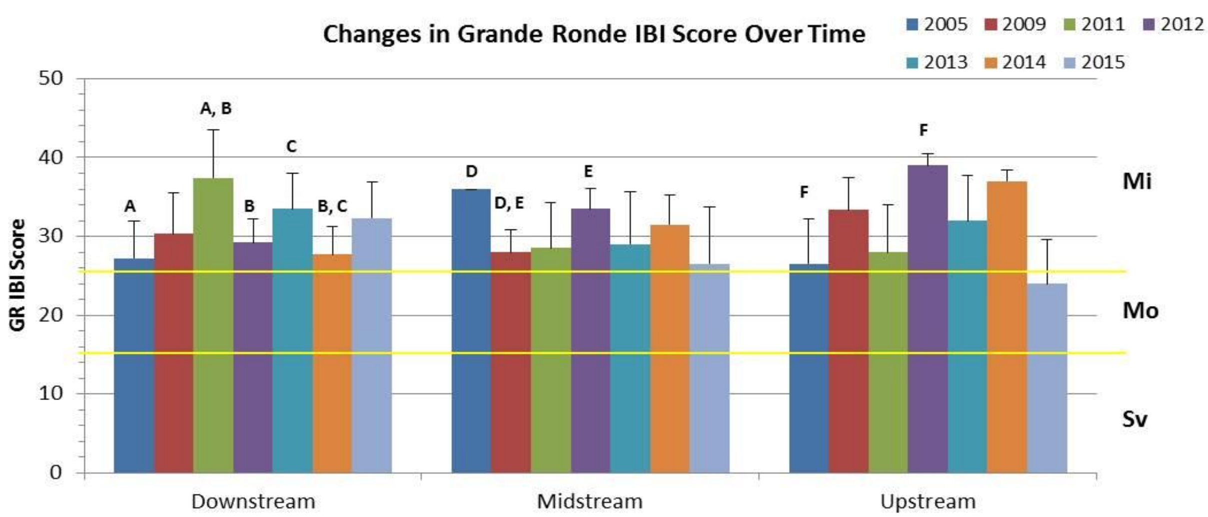


Figure 3. Grande Ronde IBI scores for Whychus Creek sampling sites.

Predator

Site O/E

PREDATOR scores were higher in 2015 compared to recent years, with 23% of the sites scoring as good, 54% as fair, and 23% as poor. This is a contrast to 2014 PREDATOR scores, where the same sites all had O/E scores reflecting poor conditions. Biological conditions reflected by PREDATOR scores have in general been worse than those reflected by IBI scores, although that may be influenced in part by the fact that PREDATOR comprises only 3 impaired condition categories apart from enriched (good, fair, poor), while the general IBI has four categories that encompass this same condition span (minimal, slight, moderate, and severe impairment). PREDATOR scores at individual sites vary annually (Table 6), but the differences generally result in little change in biological condition category, though some sites have fluctuated annually between ratings of poor and fair (Figure 4A). When considered at the reach level, mean O/E scores among downstream sites increased from 2005 to 2009, though not significantly, and remained at about the same level of fair biological condition until 2014, when the mean O/E score was significantly lower. However, in 2015, the mean O/E score among the downstream reaches increased significantly compared to the previous year, and once again reflected fair conditions.

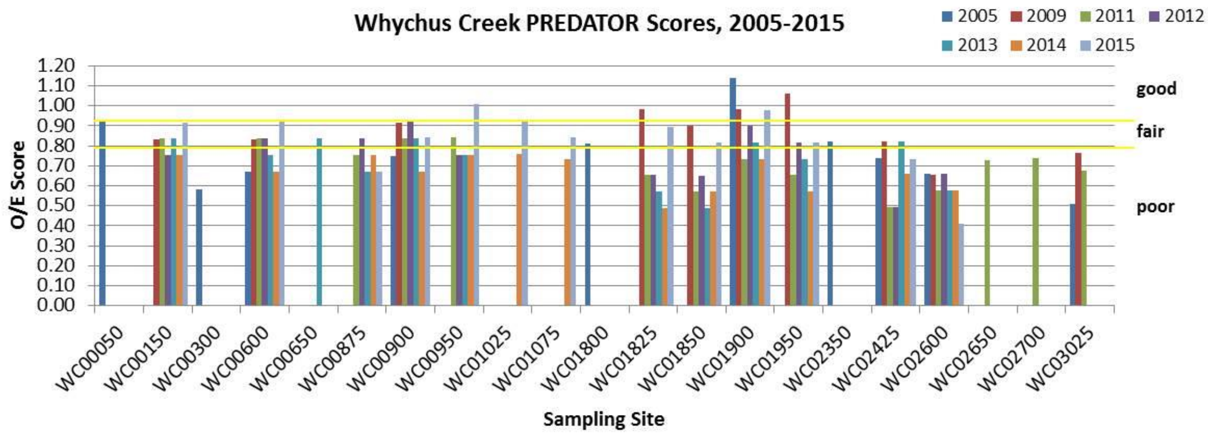
The greatest variation has been among midstream sites, with significant between-year differences in mean scores that have corresponded to good, fair, and poor conditions during this study. The highest (good) midstream reach mean O/E scores were seen in 2005; scores in subsequent years were significantly lower and reflected apparently poor biological conditions. However, the mean O/E score in 2015 was significantly greater than the means in 2011-2013, and reflected an improvement to fair biological conditions. The macroinvertebrate community may thus have recovered from perturbations caused by earlier restoration actions in this portion of the creek, and/or have improved as a result of management actions. Mean O/E scores for the upstream reach sites have consistently reflected poor biological conditions, however, and though they too vary annually there is no clear pattern and differences between years are not significant.

Table 6.
PREDATOR O/E scores. Colors indicate biological conditions that are good (orange), fair (green), or poor (blue).

Site	2005	2009	2011	2012	2013	2014	2015
WC0050	0.92	---	---	---	---	---	---
WC0150	---	0.83	0.84	0.75	0.84	0.75	0.92
WC0300	0.58	---	---	---	---	---	---
WC0600	0.67	0.83	0.84	0.84	0.75	0.67	0.92
WC0650	---	---	---	---	0.84	---	---
WC0875	---	---	0.75	0.84	0.67	0.75	0.67
WC0900	0.75	0.92	0.84	0.92	0.84	0.67	0.84
WC0950*	---	---	0.84	0.76	0.75	0.76	1.01
WC1025 (WC1100)	---	---	---	---	---	0.76	0.93
WC1075 (WC1150)	---	---	---	---	---	0.73	0.84
WC1800	0.81	---	---	---	---	---	---
WC1825	---	0.98	0.65	0.65	0.57	0.49	0.90
WC1850	---	0.90	0.57	0.65	0.49	0.57	0.81
WC1900	1.14	0.98	0.73	0.90	0.82	0.73	0.98
WC1950	---	1.06	0.65	0.82	0.73	0.57	0.81
WC2325	0.82	---	---	---	---	---	---
WC2425	0.74	0.82	0.49	0.49	0.82	0.66	0.74
WC2600	0.66	0.66	0.58	0.66	0.58	0.58	0.41
WC2650	---	---	0.73	---	---	---	---
WC2700	---	---	0.74	---	---	---	---
WC3025	0.51	0.76	0.68	---	---	---	---

*in 2012, sampling was done at RM 9.25 instead of RM 9.5

a. Individual site O/E scores across time



b. Mean PREDATOR O/E scores across time among stream reaches. Letter pairs indicate significant difference between means ($p<0.05$). Downstream = RM 0.5-11.5; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

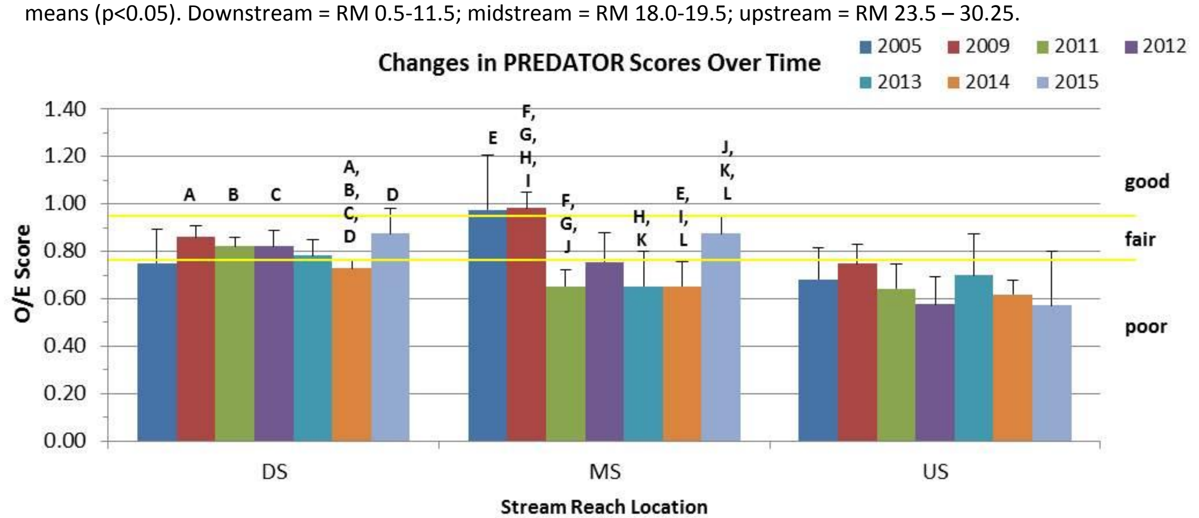


Figure 4.
PREDATOR O/E Scores for Whychus Creek sampling sites

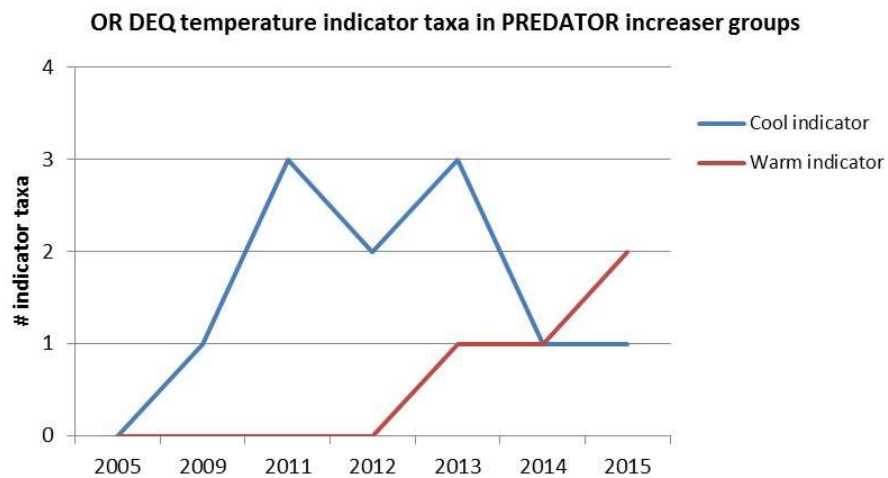
Increaser and Decreaser Taxa

The PREDATOR model output includes a sensitivity index for each taxon, calculated as the # sites at which a taxon was observed/#sites at which it was expected (S_o/S_e), such that a single sensitivity index score is generated for each taxon across the entire sampling set. Taxa with S_o/S_e >1 are considered to be "increasers" (observed more frequently than expected); taxa with S_o/S_e <1 are "decreasers" (observed less frequently than expected). Increaser and decreaser taxa for each sampling site in Whychus Creek were analyzed across all sampling years. To avoid a bias for rare taxa, increasers were identified using $S_o/S_e \geq 1.3$, while taxa with S_o/S_e <0.8 were selected as decreasers.

Increaser and decreasers included several taxa identified by OR DEQ as indicators for temperature and/or sediment conditions (Appendix B), and changes in the numbers of indicator taxa suggest a response to altered stream conditions (Figures 5 and 6). In 2005, there were no cool temperature indicators among the increaser group, but the number of cool indicator taxa rose among increasers through 2013 (Figure 5A). Conversely, among the decreaser taxa, no warm indicators were seen In 2005 and 2009, after which the number of warm indicators rose and remained high among the decreaser groups (Figure 5B). In both 2014 and 2015, the number of cool indicator taxa also rose among the decreaser groups, which may suggest a new temperature stressor.

A pattern of increased numbers of low sediment indicator taxa and decreased numbers of high sediment indicators is apparent in the increaser groups, while the opposite occurred among the decreasers, with increasing numbers of high sediment indicators and low numbers of low sediment indicators in all years except 2014 (Figure 6).

a. Number of cool and warm indicator taxa in increaser group



b. Number of cool and warm indicator taxa in decreaser group

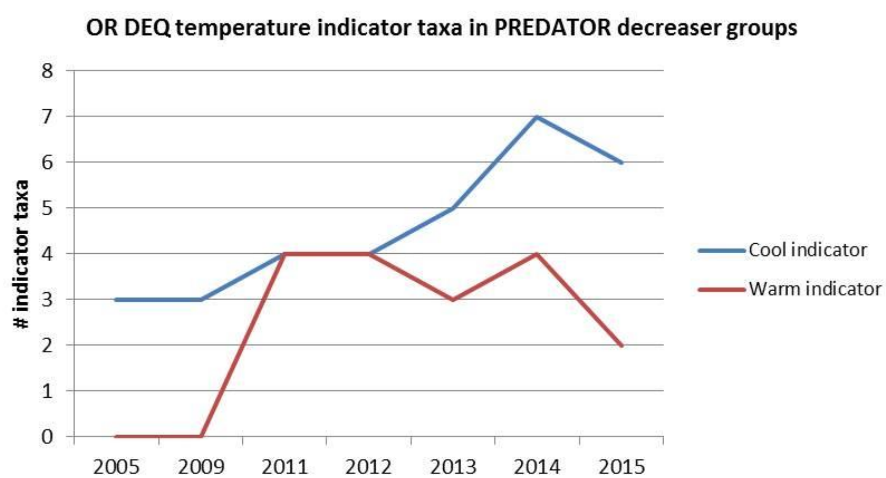
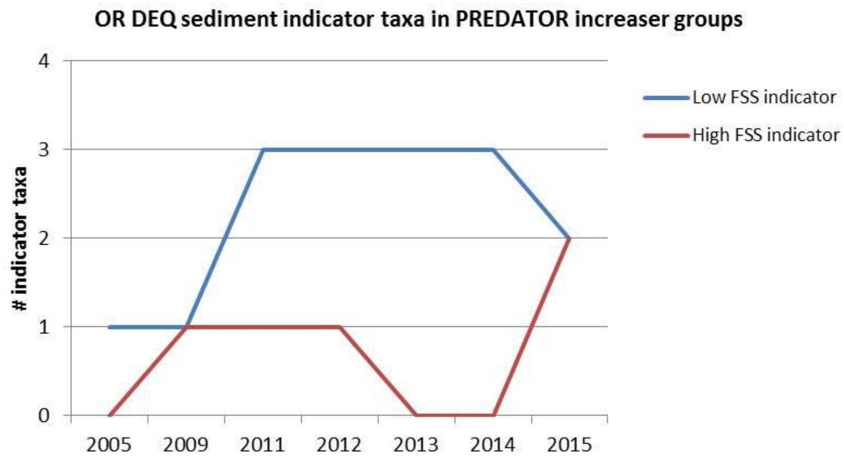


Figure 5. Indicator taxa for temperature among PREDATOR increaser and decreaser taxa groups.

a. Indicator taxa for sediment among PREDATOR increaser groups.



b. Indicator taxa for sediment among PREDATOR decreaser groups.

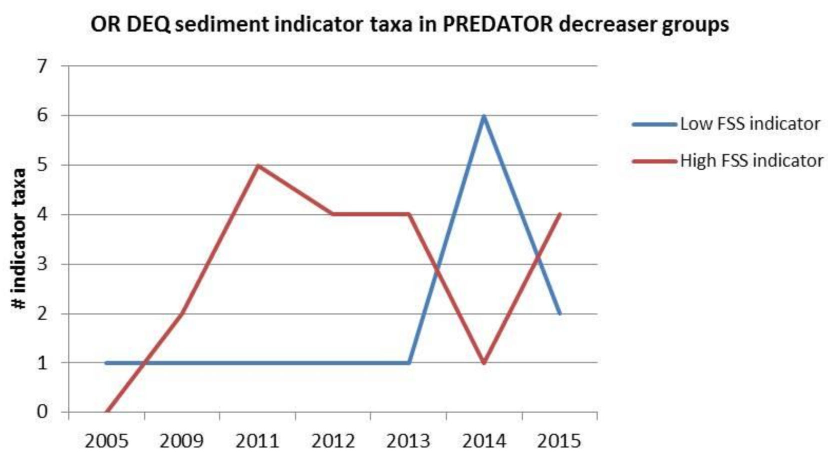
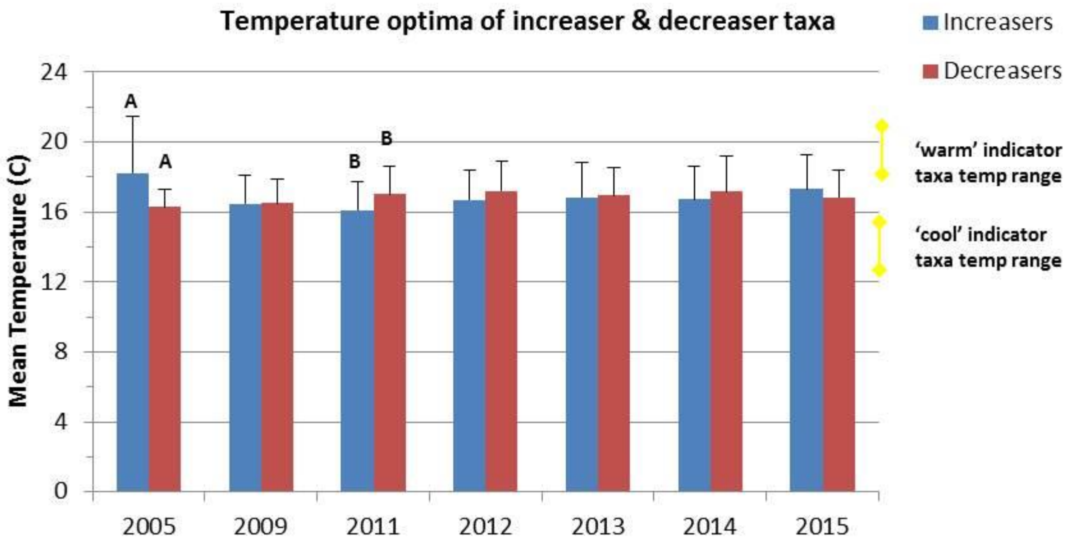


Figure 6. Indicator taxa for fine suspended sediment (FSS) among PREDATOR increaser and decreaser taxa groups.

Increaser communities tend to have lower mean temperature and sediment optima compared to decreaser communities, although differences are not significant in every year (Figure 7). The greatest differences are seen in sediment optima; the mean %FSS optima of the increaser community are lower than the decreaser community in every sampling year (Figure 7B), and this difference was significant in 2012 and 2014 ($p<0.05$) and near significant in 2011 ($p=0.06$). Temperature optima in increaser and decreaser communities have differed less; mean optima are similar within years, and no significant differences have been seen since 2011 (Figure 7A). However, the consistent pattern of higher mean sediment optima among the decreaser groups in each year suggests that stream conditions are facilitating colonization and survival of taxa with greater sensitivity to sediment levels.

a. Mean temperature optima of increaser and decreaser taxa. Letter pairs indicate significant differences (p<0.05). Temperature ranges of OR DEQ indicator taxa shown for reference.



b. Mean %FSS optima of increaser and decreaser taxa. Letter pairs indicate a significant difference (p<0.05). %FSS ranges of OR DEQ indicator taxa shown for reference.

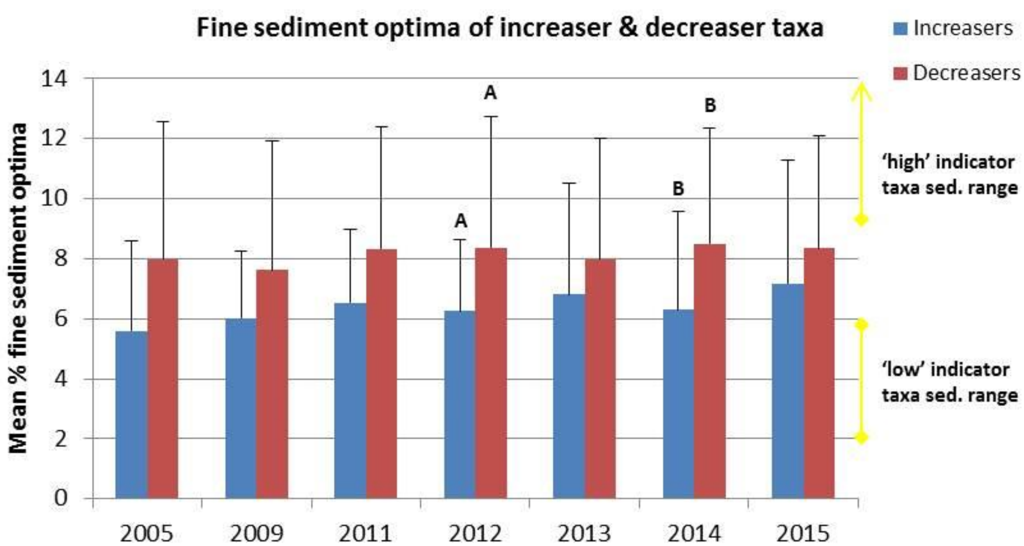


Figure 7. Mean temperature and fine sediment optima among increaser and decreaser taxa

The composition of increaser and decreaser groups varies annually, but some taxa are present consistently. *Acentrella* (small minnow mayfly), *Brachycentrus (*flatheaded mayfly), Capniidae (small winter stonefly family), and *Rhithrogena* (humpless casemaker caddisfly) are among the increaser taxa each year; all are members of the EPT (Ephemeroptera, Plecoptera, Trichoptera) group considered most sensitive to degraded stream conditions. *Rhithrogena*, Capniidae, and *Brachycentrus* have a low tolerance for organic pollution, and *Rhithrogena* is also an OR DEQ low temperature indicator taxon. Taxa that occur consistently as decreasers are more mixed and include sensitive and more tolerant taxa such as *Drunella doddsi* (a sensitive spiny crawler mayfly with low tolerance for organic pollution; DEQ cool temperature and low sediment indicator), *Heterlimnius* (riffle beetle), *Ironodes* (flatheaded mayfly; DEQ cool temperature indicator), *Lepidostoma* (casemaker caddisfly with low tolerance for organic pollution), *Malenka* (small brown stonefly with low tolerance to organic pollution), *Maruina* (moth fly), *Micrasema* (of humpless casemaker caddisfly with low tolerance for organic pollution; DEQ cool temperature indicator) and *Neophylax* (stonecase caddisfly).

CLUSTER analysis of a Bray-Curtis similarity matrix of presence/absence abundance of decreaser taxa (Figure 8) groups the 2005 and 2009 communities separately; in later years, decreaser community composition was most similar among the 2011-2013 communities and the 2014 and 2015 communities, suggesting a continuous shift in stream conditions. CLUSTER analysis of increaser taxa also separates the communities in 2005 and 2009 from all later years.

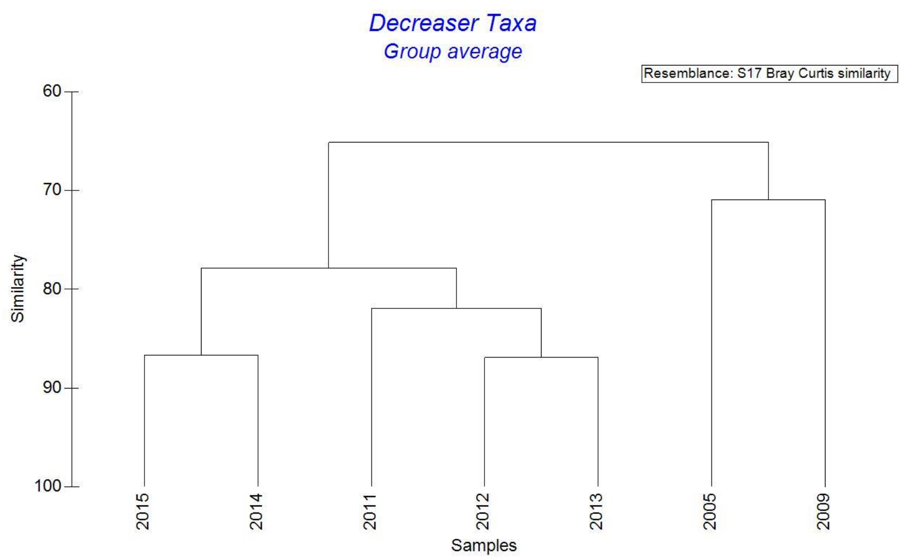


Figure 8. Similarity among PREDATOR decreaser communities

Missing and Replacement Taxa

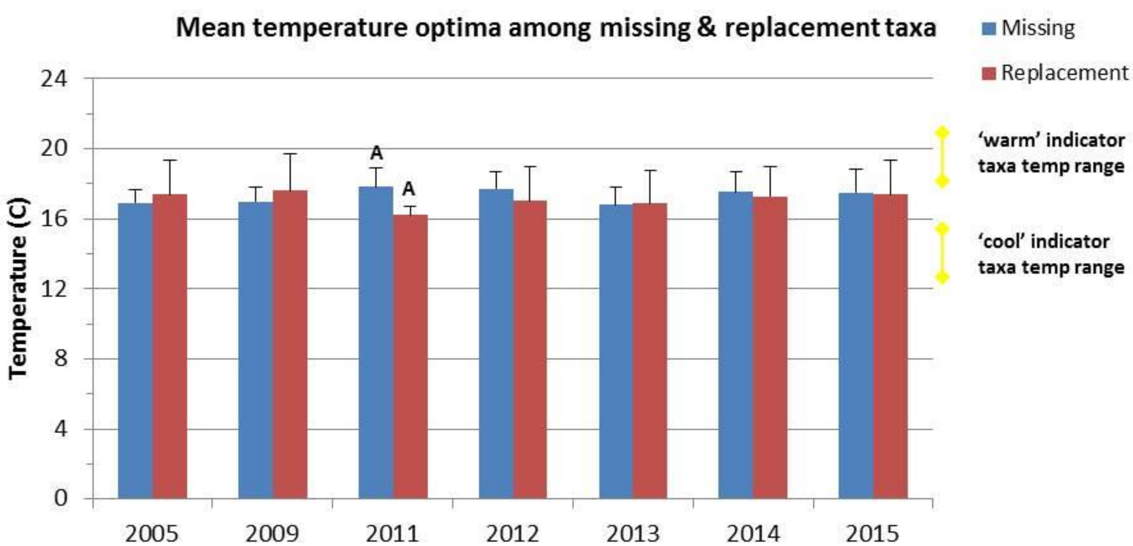
The PREDATOR model further identifies taxa expected at a site but not collected (missing taxa), and taxa that were not expected at a site but were collected in samples (replacement taxa), based on comparison to appropriate reference sites. Missing and replacement taxa are indicated for each individual sampling site. Taxa missing from a majority of expected sites (≥7) in every year included *Calineuria* (a moderately sensitive perlid stonefly), *Malenka* (small brown stonefly with low tolerance to organic pollution), Pisidiidae (a family of sediment- and pollution-tolerant fingernail clams; DEQ high sediment indicator), and Tanypodinae (a common tolerant non-biting midge group; DEQ high sediment indicator). *Malenka* and Pisidiidae were also among the decreaser taxa in each year. Taxa identified as replacements at ≥7 sites in each sampling year included *Acentrella* (a small minnow mayfly), *Atherix* (a common, tolerant watersnipe fly), *Narpus* (a common, moderately tolerant riffle beetle), *Rhithrogena* (a flatheaded mayfly with low tolerance for organic pollution; DEQ low sediment indicator), and *Serratella* (a common spiny crawler mayfly with low tolerance to organic pollution). *Rhithrogena* and *Acentrella* were also among taxa identified as increasers in each year. No new missing taxa were identified in the 2015 samples, but three replacement taxa in the 2015 samples had not been seen as replacements in any previous year: *Diphetor hageni* (small minnow mayfly; OR DEQ warm temperature indicator), *Hydroptila* (a tolerant microcaddisfly; OR DEQ high sediment indicator taxon)and Dasyheleinae (a biting midge subfamily).

CLUSTER analysis of a Bray-Curtis similarity matrix of presence/absence abundances of missing taxa showed two main groups, with the 2005 and 2009 communities most closely related to each other (91%

similarity) and then to the 2013 missing community (71% similar). The remaining years clustered into a second group, with greater similarity between the 2011 and 2012 missing community (94% similarity) and 2014 and 2015 community (89% similarity). However, the two main groups had an overall similarity of 69%, showing a high degree of commonality among missing taxa throughout all sampling years. Less overall similarity was seen among replacement communities in earlier sampling years, with the replacement communities from 2005-2011 having only 53-66% similarity. The replacement communities in recent years were more closely related, with 85% similarity between the 2014 and 2015 replacement communities and between the 2012 and 2013 replacement communities.

A comparison of mean temperature and sediment optima among missing and replacement communities reveals a pattern similar to that seen for decreasers and increasers (Figure 9). In 2005 and 2009, mean temperature optima were higher among replacement taxa, but in 2011, the mean temperature optima of the replacement community was significantly lower than the missing community, and in subsequent years the temperature optima of the two groups are similar. A stronger trend occurs for %FSS optima, which are significantly lower in replacement versus missing communities in every year except 2009 (and the difference in 2009 is close to significant, at p=0.056).

a. Mean temperature optima of missing and replacement taxa. Letter pairs indicate a significant difference (p<0.05). Temperature ranges spanned by ORDEQ indicator taxa shown for reference.



b. Mean %FSS optima of missing and replacement taxa. Letter pairs indicate a significant difference ($p<0.05$). The OR DEQ indicator taxa range is shown for reference.

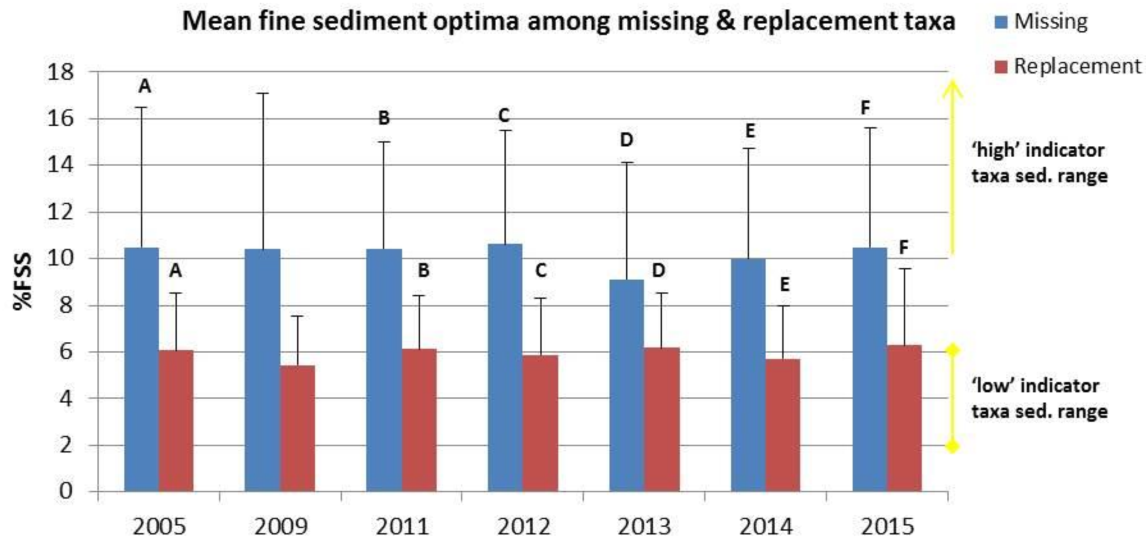


Figure 9. Differences in temperature and fine sediment tolerances among missing and replacement taxa

Taxonomic and Ecological Trait Analysis

<u>Community analysis</u>

Community composition

Sample organismal abundance has been high enough that the target subsampling count of 500 was achieved for most samples in most years; samples with lower abundances have generally come from more upstream sites, which is not unusual for small cold headwater streams (Crunkilton & Duchrow 1991, Lillie *et al*. 2003). From 2005-2013, the 500 organism target was not attained for only one to three samples from within a given year. In 2015, however, abundances seemed lower overall, as not only was the 500-organism subsampling target not met for four of the 13 composite site samples, but these samples had unusually low numbers of invertebrates, ranging from 67 to 264. The reason for this is not known, and differences in sampling technique between volunteers cannot be ruled out, but these lower abundances are likely influenced by the abnormally hot dry weather in 2015 and the occurrence of significant recent restoration-related disturbances at a flood plain project site (RM26; composite sample contained 67 organisms).

Seventy-two unique taxa were collected among all sites in 2015, which is in the same range as other years (76-83 taxa per year in 2005–2014) but is the lowest number of taxa collected in a single sampling year. Despite lower overall abundances and diversity, three new taxa were added to the list in 2015, bringing the number of unique taxa collected among all sampling sites since 2005 to 141 (see Appendix C for taxa list). New taxa not seen in previous years were *Labiobaetis*, *Acentrella insignificans*, and *Attenella delantala*. *Labiobaetis* is a small minnow mayfly genus that is widely distributed across a variety of lotic habitats; it tends to occur in low abundance when present (only one individual was found at a single site in 2015) and is thus not unusual as an occasional record in longer-term data sets (Mike Cole, pers. comm.). *Acentrella insignificans* is also in the small minnow mayfly family; one to two individuals of this species were present at three sampling sites in the lower reaches in 2015, and additional species in this genus were present in samples in other years. A single individual of *Attenella* *delantala,* a spiny crawler mayfly, was found at one sampling site. Both *Labiobaetis* and *Acentrella insignificans* have low %FSS optima (3% and 4%, respectively), in keeping with the trend for low sediment optima seen among increaser and replacement taxa.

The rate of addition of new taxa to the species list has slowed dramatically. In 2009, 35 new taxa were found among all sites, 69% of which were in the sensitive EPT group. This dramatic increase in 2009 occurred in part because four years had passed since the first sampling was done and restoration activities that increased stream flows had been implemented in the interim, and because several groups that had been left at genus in 2005 were identified to species in 2009 (and later years). Fewer new taxa were seen in every subsequent year, but except for 2014, the majority of new species are EPT, with 10 new taxa in 2011 (80% EPT), three in 2012 (100% EPT), seven in 2013 (71% EPT), five in 2014 (20% EPT), and three in 2015 (100% EPT). This suggests a larger initial shift in the macroinvertebrate community in response to restoration activities that favored colonization by sensitive taxa, with increased stabilization in recent year.

Agreement between replicate samples

In each year, duplicate samples are taken at two sites by Xerces staff as a quality control measure to assess operator differences between newly-trained volunteers and experienced stream samplers. A CLUSTER analysis showed that in 2015, as in almost every previous year, each duplicate sample community was most similar in composition to the sample taken by volunteers at the same site. This provides assurance that volunteers correctly implement sampling protocols, and that differences seen between sites and years are due more to changes in the macroinvertebrate community than differences among practitioners.

Community similarity analysis

CLUSTER analysis and MDS ordination of the 2015 dataset revealed greatest macroinvertebrate community similarity among samples taken in the same reaches (Figure 10), with downstream (46% similar), mid-reach (49% similar), and upstream (55% similar) sites forming three separate groups, as expected based on the River Continuum Concept (Vannote et al., 1980). Similar relationships are evident in all years among sites in the same sampling reaches, with average community similarities of 66% among all downstream and mid-reach sampling sites, and 58% among all upstream sampling sites. The greatest between-year differences continue to be seen for the 2005 sampling community compared to all other years (Figure 11).

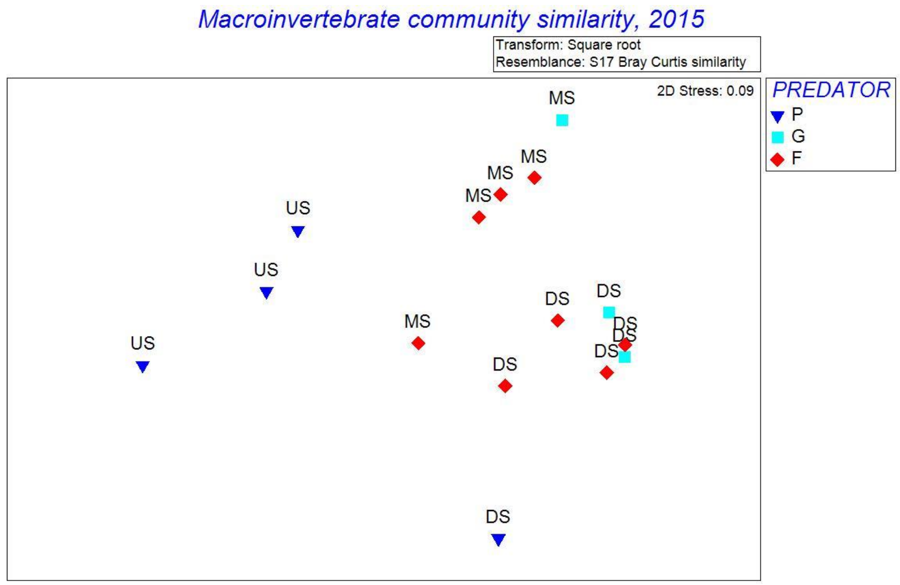


Figure 10. MDS ordination of macroinvertebrate communities in Whychus Creek, 2015. Downstream (DS) reach = RM 0.5 – 11.5; mid-stream (MS) = RM 18.0 – 19.5; upstream (US) reach = RM 23.5 – 30.25.Site PREDATOR score (Poor, Fair, Good) is also indicated.

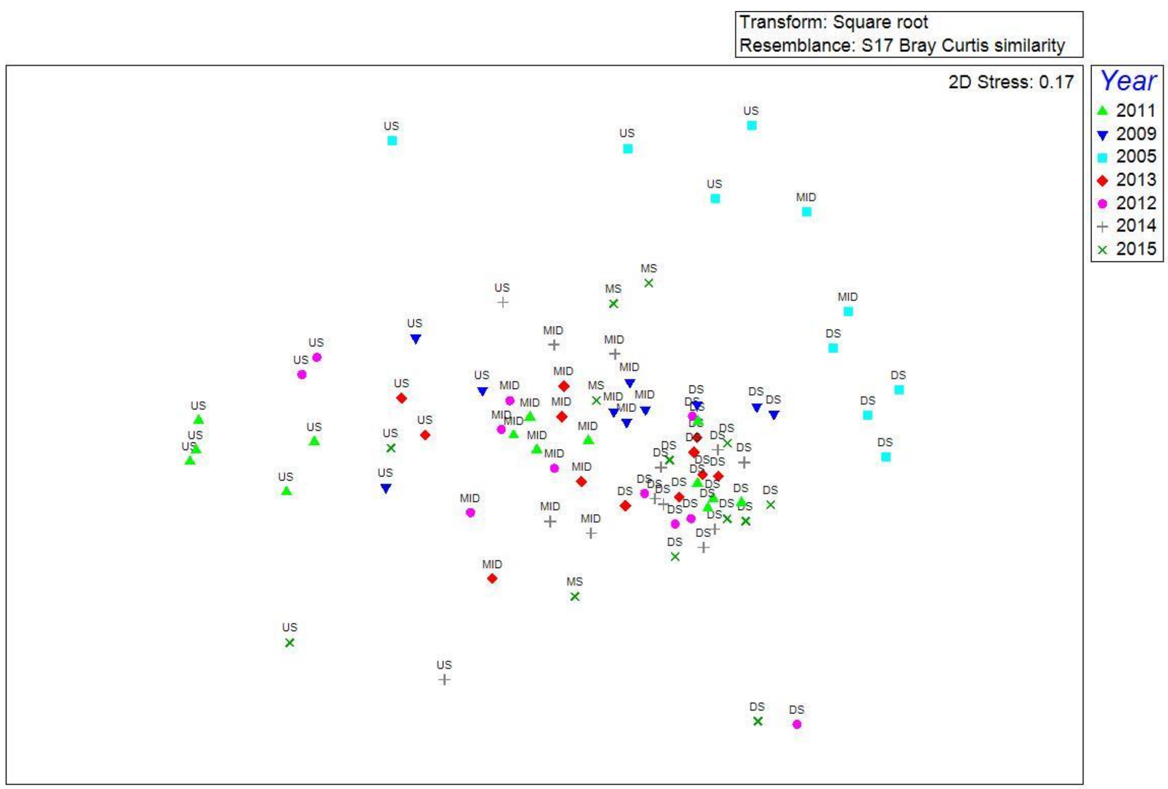


Figure 11. MDS ordination of macroinvertebrate communities in Whychus Creek, 2005-2015. Downstream (DS) reach = RM 0.5 – 11.5; mid-stream (MS) = RM 18.0 – 19.5; upstream (US) reach = RM 23.5 – 30.25. Site PREDATOR score (Poor, Fair, Good) is also indicated.

Downstream sites

MDS ordination of downstream sampling sites showed a distinct separation between the 2005 community compared to later years (Figure 12). The lowest average between-year dissimilarity (35%) was seen in 2013 and 2014, and the communities in 2005 and 2012 had the greatest average dissimilarity (55%). The macroinvertebrate community in 2009 was the most similar to the 2005 community (48% dissimilarity), while those in all subsequent years were 50-55% dissimilar to the 2005 community. The downstream macroinvertebrate community in 2015 was 52.4% dissimilar to the 2005 community, and the taxa that contributed most to the differences were *Baetis* (common widespread genus of small minnow mayfly; includes species that are OR DEQ cool temperature and low sediment indicator taxa; more abundant in 2015), *Brachycentrus* (sensitive humpless case-maker caddisfly; more abundant in 2015), and *Zaitzevia* (tolerant riffle beetle; OR DEQ warm temperature and high sediment indicator; more abundant in 2005). The downstream communities in 2014 and 2015 were only 43% dissimilar, due primarily to differing abundances of *Simulium* and *Baetis* (both more abundant in 2014), as well as *Brachycentrus* (more abundant in 2015).

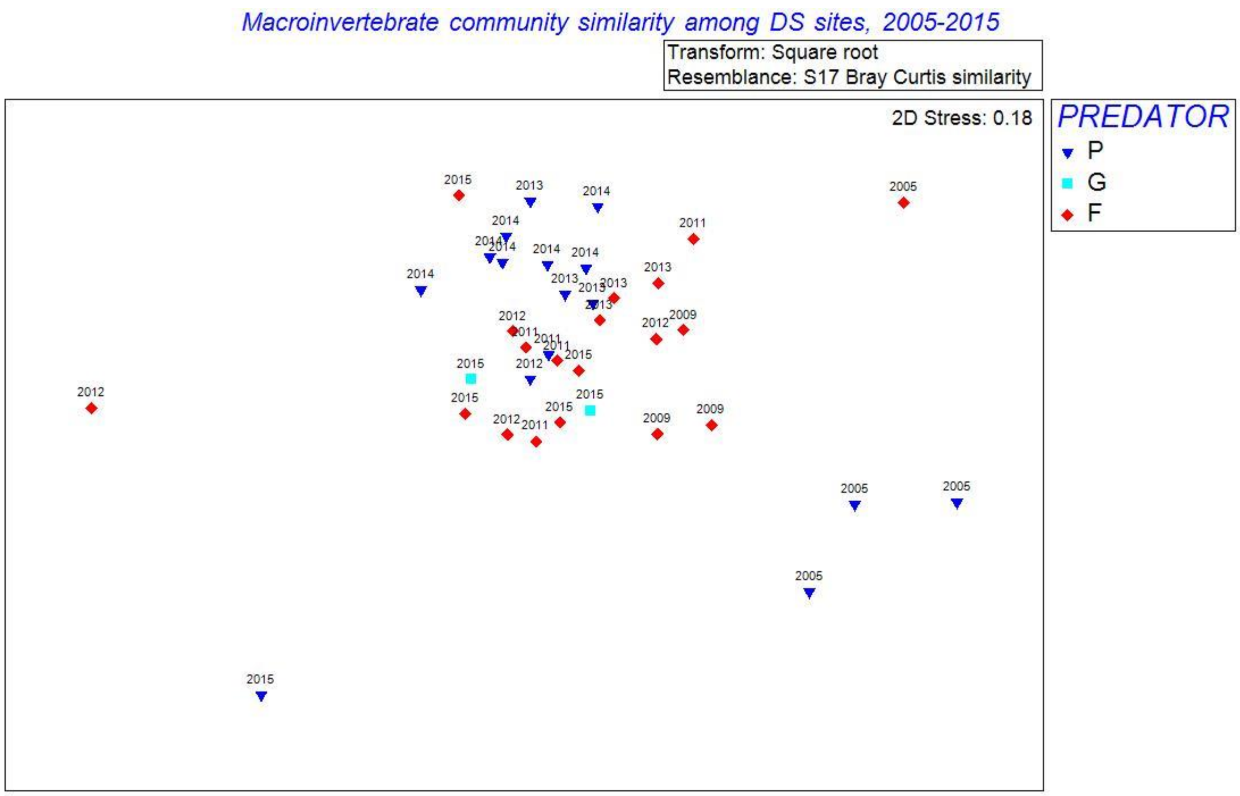


Figure 12. MDS ordination of macroinvertebrate communities among downstream sampling sites (RM 0.5 – 11.5) in Whychus Creek, 2005-2015. Site PREDATOR score (Poor, Fair, Good) is also indicated.

Mid-reach sites

MDS ordination of mid-reach sampling sites also separates the 2005 community from later years (Figure 13). The lowest between-year community dissimilarity was seen for 2011 and 2012 (37% average dissimilarity), and the greatest average dissimilarities were seen between the community in 2005 versus each year in the 2011-2014 span (62-70% average dissimilarity).

Taxa that contributed the most to differences with the 2005 community were similar in different years, and included Baetis (common genus of small minnow mayfly; lower abundance in 2005), Zapada (relatively sensitive genus of small brown stonefly; greater abundance in 2005); Simulium (moderately tolerant cosmopolitan genus of black fly; greater abundance in 2005); and Chironominae (diverse subfamily of non-biting midge with many tolerant genera; greater abundance in 2005). Taxa that contributed most to differences between the 2014 and 2015 mid-reach communities, which were 48% dissimilar, also included Baetis (more abundant in 2014) as well as Chironominae and Oligochaeta, tolerant groups that were more abundant in the 2015 community, which may indicate stressors related to abnormally low precipitation and high temperatures in that year.

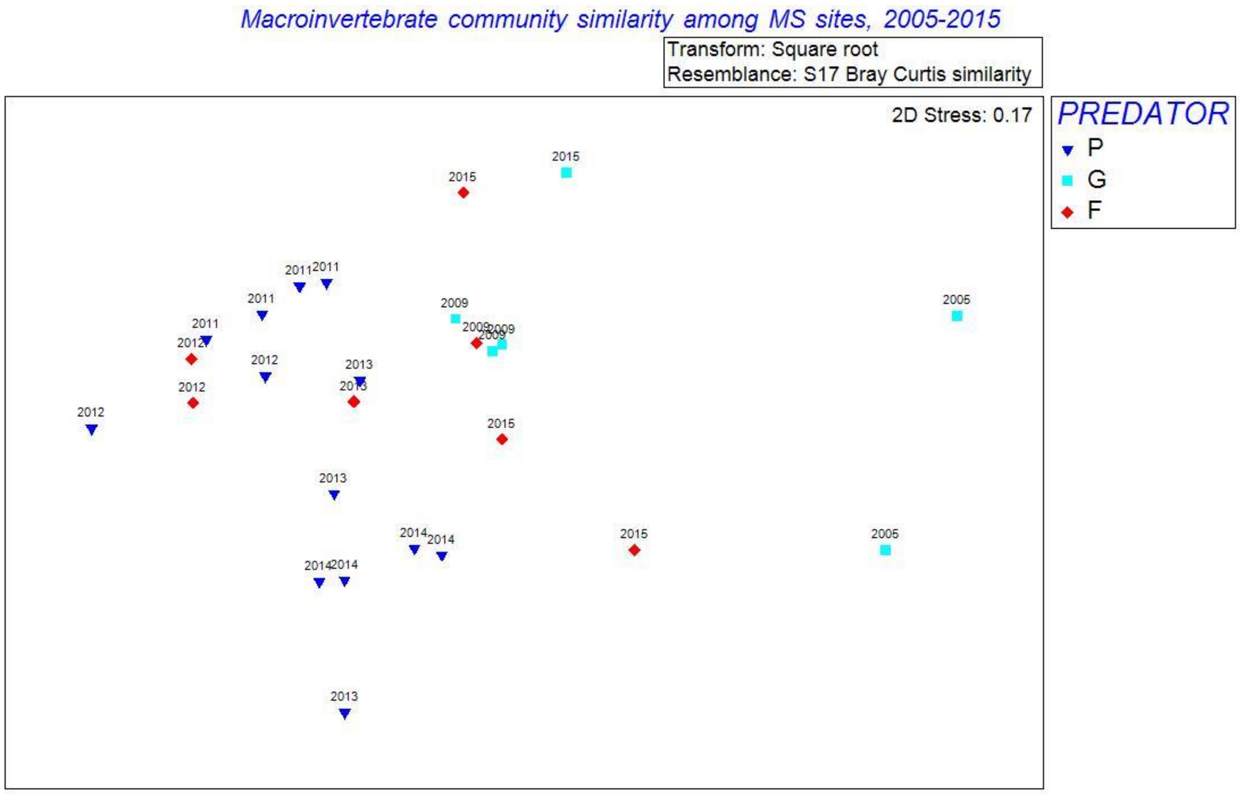


Figure 13. MDS ordination of macroinvertebrate assemblages at all mid-stream reach sampling sites (RM 18.25 – 19.5). Site PREDATOR score (Poor, Fair, Good) is also indicated.

Upstream sites

MDS ordination of upstream sampling sites separated the 2005 community from all later years (Figure 14). The lowest average between-year dissimilarity was in 2009 and 2013 (39%), and the greatest average dissimilarities were seen between the upstream community in 2005 and each year in the 2011-2015 span (64-75% average dissimilarity). Taxa that contributed most to differences with the 2005 community were similar between different years; those that were consistently more abundant in the 2005 community included the tolerant non-biting midge subfamilies Orthocladiinae and Chironominae, and taxa that were more abundant in subsequent years included more sensitive groups and/or taxa more associated with lotic waters, such as Rhithrogena (flatheaded mayfly genus with low tolerance for organic pollution; DEQ low sediment indicator), Simulium (moderately tolerant genus of black fly), Prosimulium (black fly genus; OR DEQ cool temperature indicator), Baetis (common small minnow mayfly), and Suwallia (sensitive genus of green stonefly).

The upstream macroinvertebrate communities present in 2014 and 2015 had an average dissimilarity of 62%, due primarily to differing abundances of Baetis and Orthocladiinae, both of which were more abundant in the 2014 samples, as well as Rhithrogena, which had greater abundances in 2015 samples.

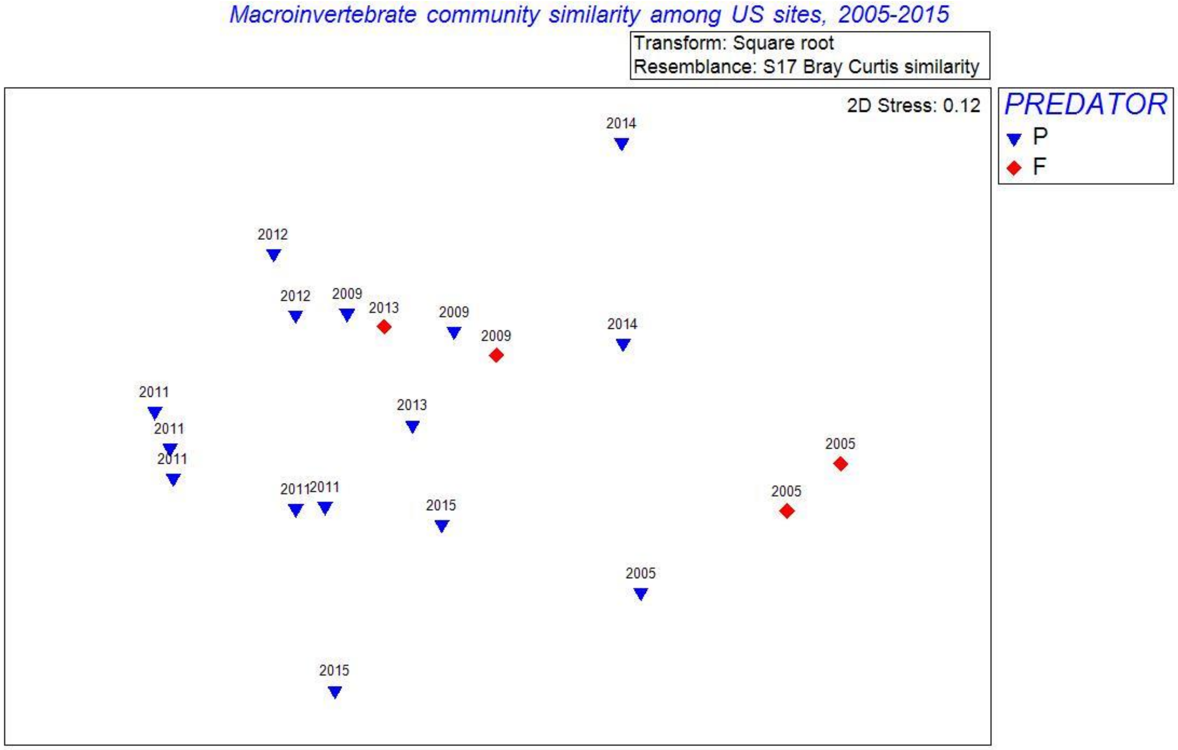


Figure 14. MDS ordination of macroinvertebrate assemblages at all upstream sampling sites (RM 23.5 – 30.25). Site PREDATOR score (Poor, Fair) is also indicated (no upstream site had an O/E score reflecting Good condition in any year).

Changes in individual IBI metrics

The OR DEQ IBI comprises different classes of metrics: richness (# taxa), tolerance (# or % tolerant or sediment-sensitive taxa), and composition (dominance of top taxon). Total IBI scores vary among sites and years, but since these scores are scaled from raw values, it can be informative to examine changes in individual metrics at the reach level.

Diversity measures

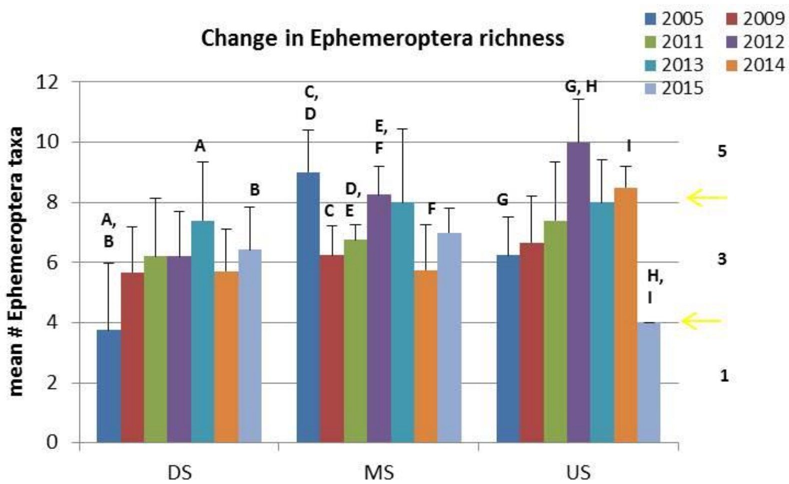
The first four metrics in the IBI are based on the rationale that a less disturbed, healthier stream system has more biodiversity with higher overall taxa richness (Norris & Georges, 1991; Barbour et al., 1996) and greater diversity of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera). The EPT are a focus of bioassessment metrics as they are widely considered to be the most sensitive to changes in flow, temperature, and sediment (although individual genera differ in their sensitivities), and they may be considered separately or as a group (EPT) in bioassessment.

Changes in taxa richness are not pronounced in most reaches (Figure 15). Mean richness oscillates between years, but differences are rarely significant. The greatest variation occurred among upstream reach sites, and diversity in 2015 was significantly lower than most previous years. Other years have seen significant increases and decreases in mean richness among upstream sites, and this decrease in 2015 likely reflects substantial disturbance both in-stream and in the riparian zone that occurred as part of the Whychus Floodplain project at RM26, as it was not apparent at downstream or mid-reach sites.

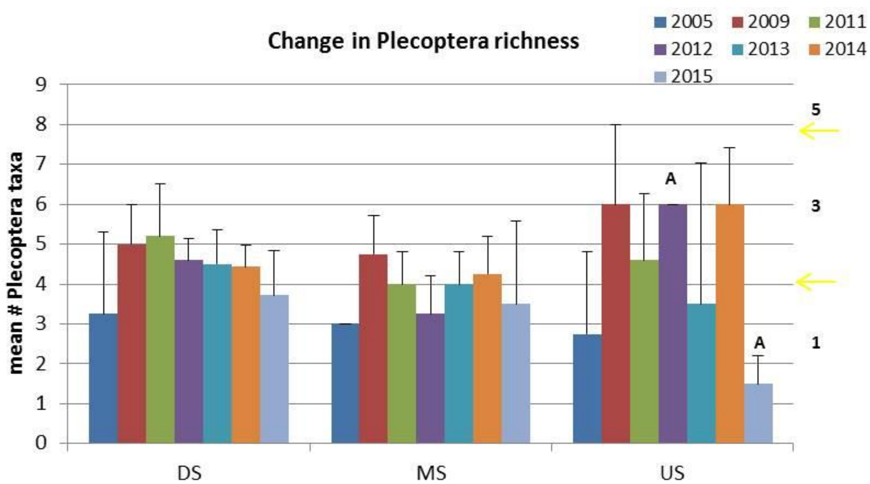
Figure 15. Changes in diversity metrics among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (P<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25. Ranges for IBI scoring (5, 3, 1) are indicated.

EPT richness was examined for each order and for EPT as a group (Figures 16 and 17). An apparent trend toward increased mayfly diversity at upstream and downstream sites continued in 2015 for the downstream reach, where the mean number of Ephemeroptera taxa remained higher than in several previously years and was significantly greater than in 2005 (Figure 16A). However, the disturbance of upstream sampling sites in 2015 resulted in a dramatic decrease in mean number of Ephemeroptera taxa among these sites. Mayfly diversity has fluctuated most among mid-reach sites, and after a significant decrease in mayfly richness among these sites in 2014, the number of taxa increased again in 2015 to levels close to those seen in 2012 and 2013. Mean stonefly diversity has generally been greater among all reaches since 2005, although these differences have not been significant (Figure 16B). The greatest variation occurred among upstream sampling sites, but until 2015, which saw the lowest mean number of stonefly taxa among upstream sites since the study began, these differences have not been significant. The mean number of caddisfly taxa has oscillated the most between sites and years, with no clear trends emerging (Figure 16C).

a. Changes in Ephemeroptera richness



b. Changes in Plecoptera richness



c. Change in Trichoptera richness

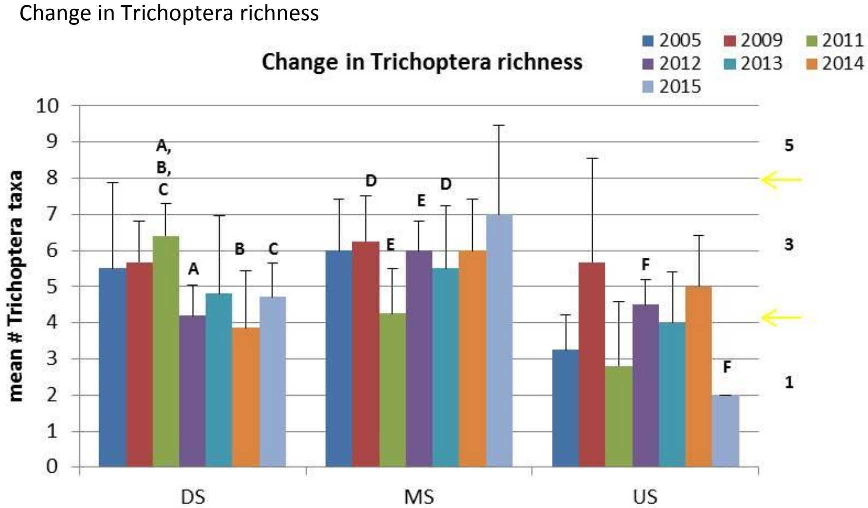
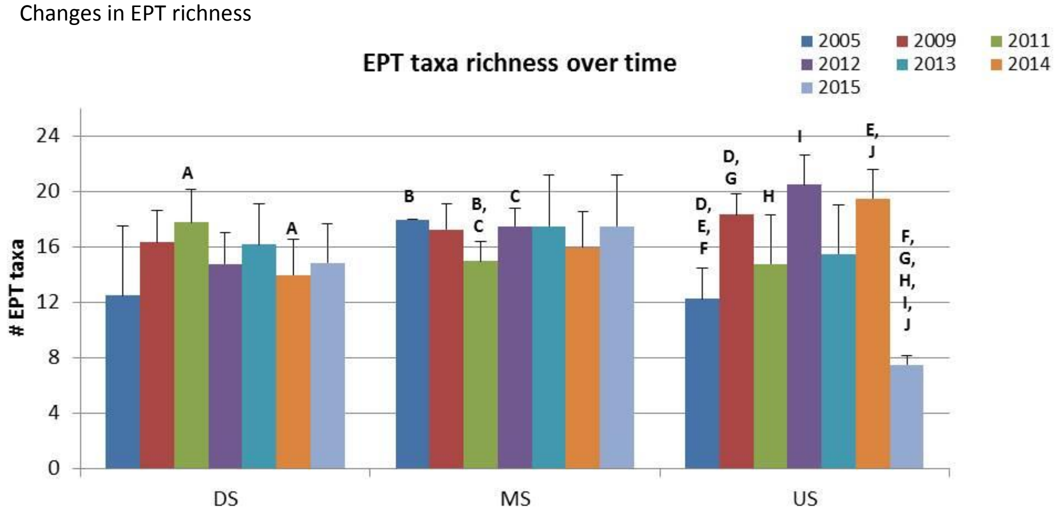


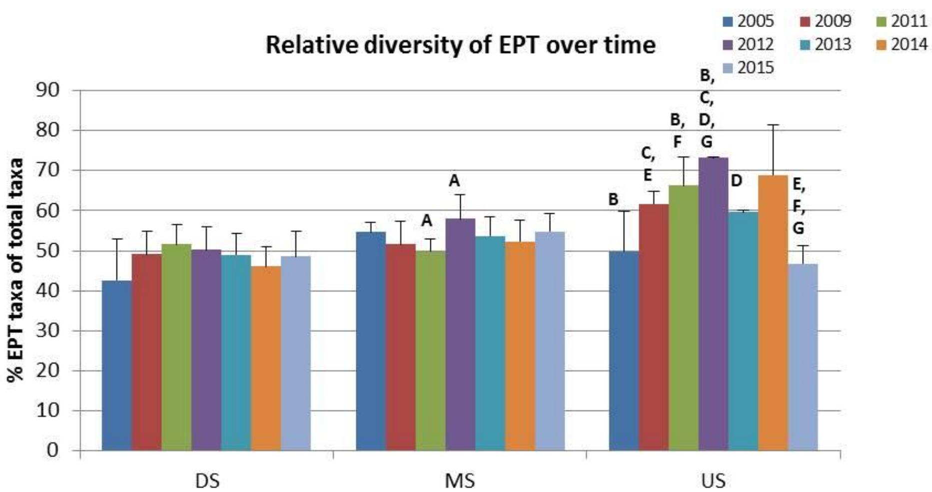
Figure 16. Changes in richness of Ephemeroptera, Plecoptera, and Trichoptera among Whychus Creek sampling sites. Letter pairs indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25. Ranges for IBI scoring (5, 3, 1) are indicated.

Total EPT richness and relative diversity have changed little in downstream and midstream reaches (Figure 17A), but a steady increase in EPT taxa among upstream sites was sharply reversed in 2015. The proportion of all taxa comprised of EPT also rose steadily among upstream sites for several years, but this trend was reversed in 2015 (Figure 75B). The relative abundance of EPT individuals among samples has increased overall among all reaches; this increase has been most sustained in downstream and upstream reaches, with more annual variation in mid-reach sites (Figure 17C).

a. Changes in EPT richness



b. Changes in EPT relative diversity



c. Changes in EPT relative abundance

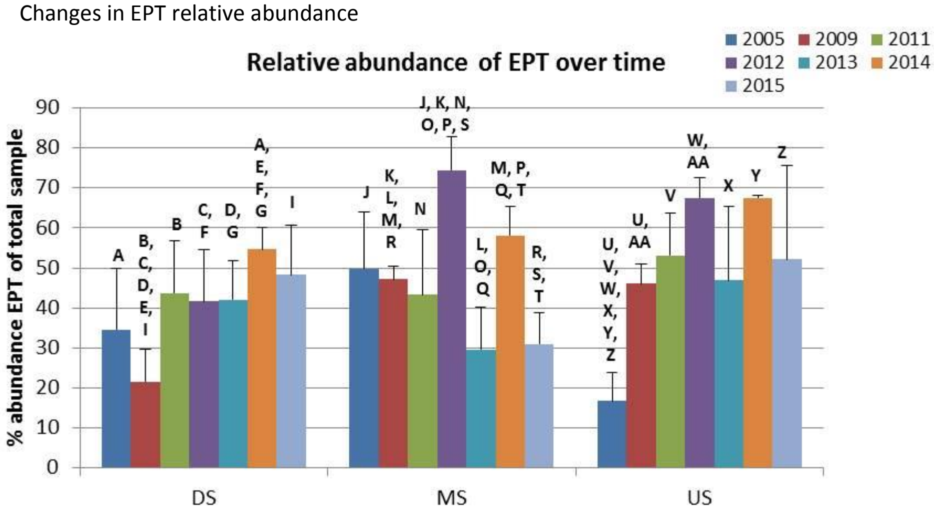


Figure 17. Changes in EPT (Ephemeroptera, Plecoptera, Trichoptera) diversity and relative abundance among Whychus Creek sampling sites. Letter pairs indicate significant difference ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Composition measures

The % dominance of the top (most abundant) taxon is a composition metric related to diversity. A healthy system is expected to have a more balanced composition, and a large abundance of a small number of taxa is considered indicative of impaired conditions and environmental stressors, as the macroinvertebrate community becomes dominated by one or a few more tolerant groups (Plafkin *et al*., 1989; Barbour *et al*., 1999). In the OR DEQ IBI, the most abundant taxon at a site must comprise less than 20% of the total abundance to receive the highest scaled score.

The proportion of the community dominated by the most abundant taxon shows a U-shaped pattern among downstream and upstream sites (Figure 18), decreasing from 2005-2011 then increasing 2013-2015, while the proportion of the dominant taxon increased at mid-reach sites though 2013 and then dropped again in recent years. The identity of the dominant taxon among sampling reaches has changed as overall macroinvertebrate community composition has changed, with the most noticeable shift occurring among downstream sampling sites. In 2005, *Glossosoma* (sensitive saddle case-maker caddisfly; OR DEQ low sediment indictor) and *Zaitzevia* (tolerant riffle beetle; OR DEQ indicator for warm temperatures and high sediment) were the dominant taxa at downstream sites, but in 2009 and 2011, dominant taxa consisted primarily of different common cosmopolitan taxa of non-biting and biting midges (Chironomidae and Ceratopogonidae, respectively), as well as Oligochaeta (disturbance- and sediment-tolerant aquatic earthworms; DEQ high sediment indicator). In 2013-2014, dominant taxa were comprised mainly of more lotic-associated taxa such as *Simulium* (common and moderately tolerant black fly) and *Baetis tricaudatus* (common cosmopolitan small minnow mayfly), and in 2015 they were joined by two caddisflies: *Brachycentrus* (humpless case-maker caddisfly sensitive to organic pollution) and *Hydropsyche* (common free-living caddisfly tolerant of organic pollution; warm temperature indicator).

Dominant taxa among mid-reach sites also changed across time. *Zapada* (small brown stonefly sensitive to organic pollution), Oligochaeta, and Orthocladiinae (subfamily of non-biting midge common in rock and gravel substrates) were the common dominant taxa in 2005-2009 among mid-reach sites, but from 2011-2014, *Baetis tricaudatus* (small minnow mayfly; DEQ low sediment indicator) and *Simulium* dominated the communities, and in 2015, *Simulium* was joined by the first instance of Tanytarsini as a dominant taxon. Tanytarsini is a tribe of non-biting chironomid midge in the subfamily Chironominae, a tolerant cosmopolitan group found in all types of aquatic habitats in sand and silt sediment (Merritt et al.; Pinder, 1986). The dominant taxa among upstream sampling sites have been more similar across time. Orthocladiinae was a dominant group in every sampling year except 2015, and Simulium and *Rhithrogena* (a flatheaded mayfly genus with low tolerance for organic pollution; low sediment indicator) occur in multiple years.

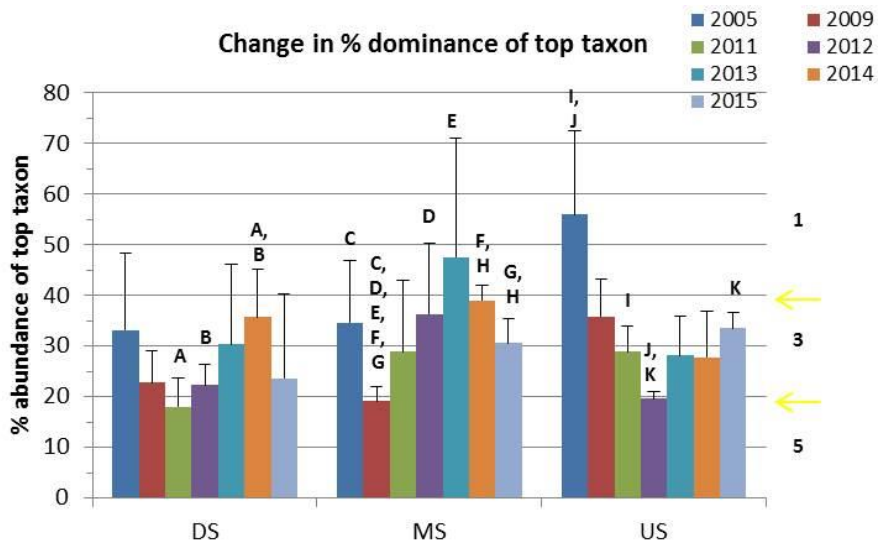


Figure 18. Dominance of the top taxon in Whychus Creek. Ranges for IBI scoring (5, 3, 1) are indicated. Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

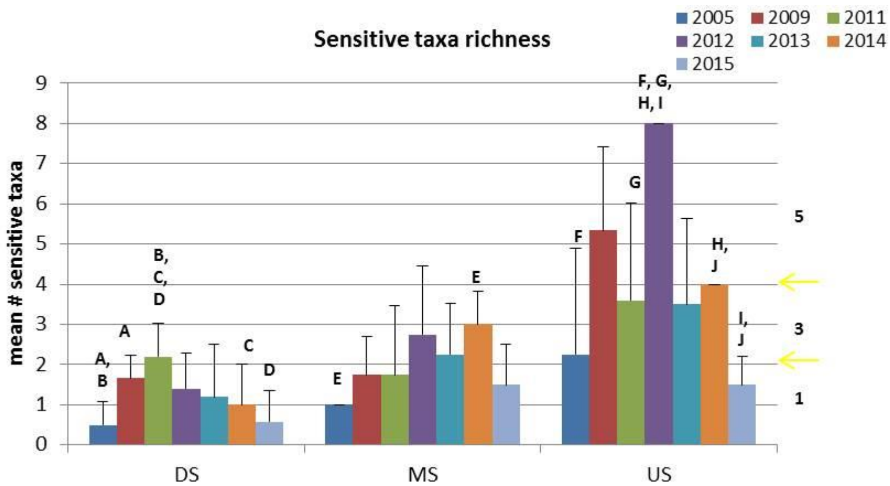
Sensitivity/tolerance measures

The remaining metrics in the IBI reflect diversity and dominance of groups that are sensitive to or tolerant of warm temperatures and increased sediment loads and include # of sensitive and sediment-intolerant taxa, %tolerant taxa (tolerant to high levels of disturbance and pollution) and sediment-tolerant taxa, and sample weighted mean of the MHBI (Modified Hilsenhoff Biotic Index) value. MHBI reflects tolerance for organic pollution (Hilsenhoff, 1987); values range from 0-10, with lower values indicating greater sensitivity. Tolerant taxa occur in both healthy and impaired habitats, but their ability to persist and even thrive under conditions with low dissolved oxygen, high turbidity, or heavy siltation means that as stressors increase and sensitive taxa drop out of the community, their relative abundance in the community increases.

Sensitive taxa diversity is highest among upstream sampling sites (Figure 19A) and increased steadily from 2005 to 2012, but sensitive taxa diversity at upstream sampling sites has decreased in recent years, and in 2015 the mean number of sensitive taxa was significantly lower than in both 2012 and 2014. Mid-reach sampling sites also showed a sustained increase in mean number of sensitive taxa since 2005, and although diversity was lower in 2015, the difference was not significant. Downstream sites have the lowest sensitive taxa diversity overall, and though sensitive taxa diversity rose among downstream sites from 2005 to 2011, this was followed by a continued decrease, and the mean number of sensitive taxa among downstream sites was significantly lower in 2014 and 2015 compared to 2011.

The mean numbers of sediment-intolerant taxa have shown an overall downward trend across all reaches over time, which does not reflect the observed pattern of lower sediment optima among the total community and among replacement and increaser taxa (Figure 19B). However, mean values in each year have had large standard deviations, and since the overall range of sediment-intolerant taxa is very limited in each year, ranging from only 0-2 among all sites, this metric has limited information content.

a. Number of sensitive taxa



b. Number of sediment-sensitive taxa

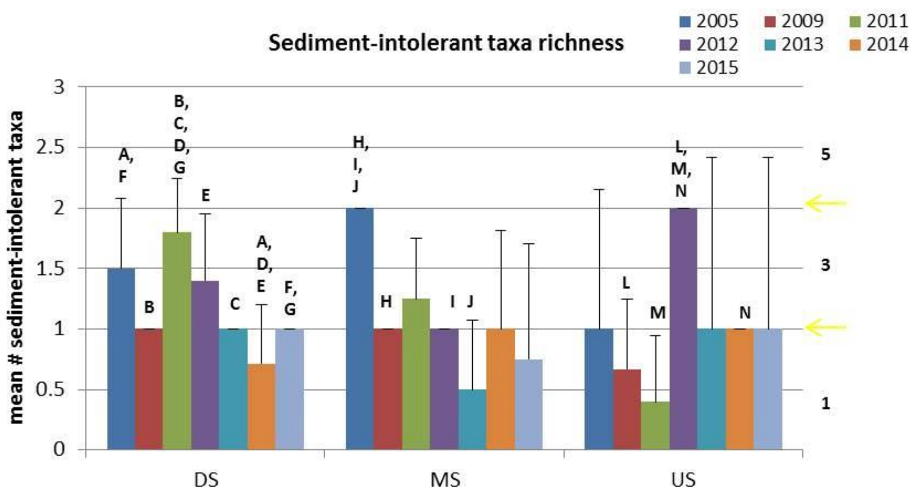
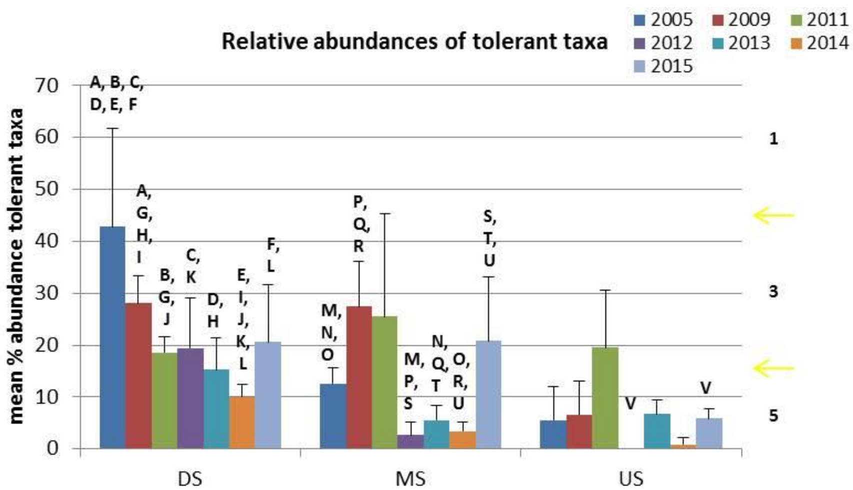


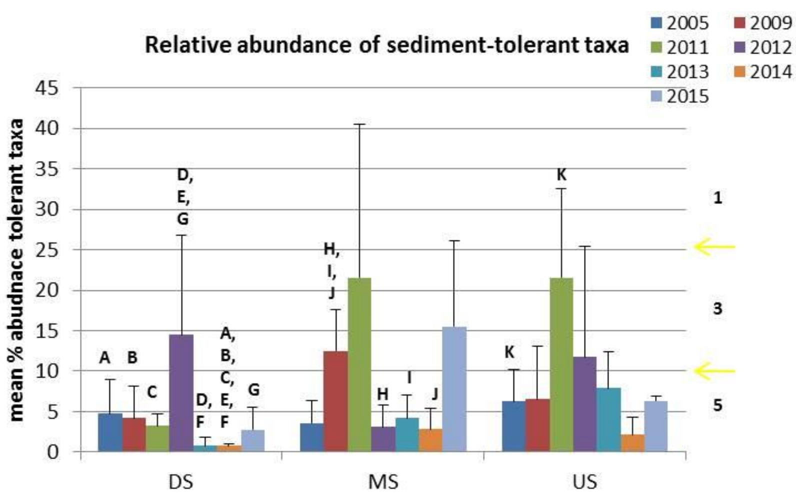
Figure 19. Richness of sensitive taxa among Whychus Creek sampling sites. Ranges for IBI scoring (5, 3, 1) are indicated. Letter pairs indicate significant difference between means. Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Greater changes have occurred among tolerant taxa in the sampling communities. Relative abundance of both tolerant and sediment-tolerant taxa decreased overall among all sampling reaches (Figure 20A, B), suggesting a response to improved stream conditions. However, in 2015 there were significantly greater proportions of tolerant taxa among all sampling reaches; relative abundance of sediment-tolerant taxa was higher as well but not significant. Mean assemblage MHBI in 2015 was similar to or lower than values seen in previous years (Figure 20C), and while mean community MHBI has varied across time, values have remained below or close to the range that receives the highest scaled score in the IBI (MHBI <4).

a. Changes in relative abundance of tolerant taxa



b. Changes in relative abundance of sediment-tolerant taxa



c. Changes in weighted mean of assemblage MHBI

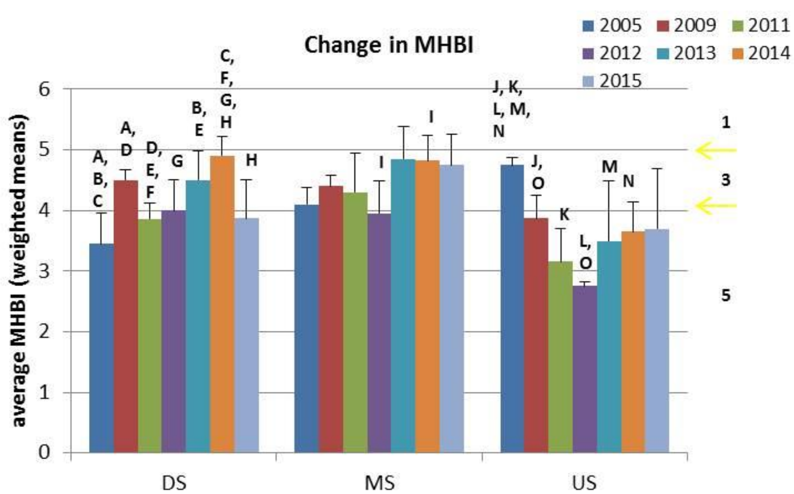


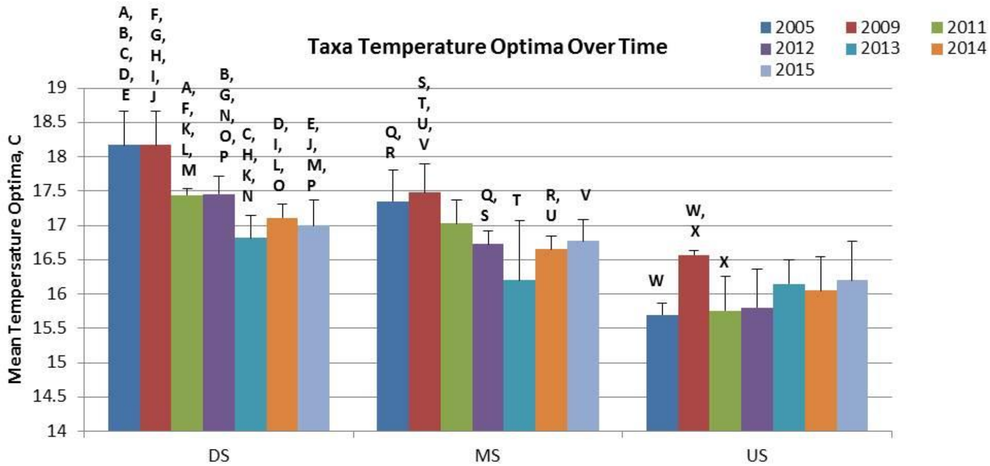
Figure 20. Changes in tolerant taxa measures among Whychus Creek sampling sites. Ranges for IBI scoring (5, 3, 1) are indicated (note that these are negative metrics, i.e. a higher raw value receives a lower scaled IBI score). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Community temperature and sediment optima

The weighted mean of temperature and sediment optima of macroinvertebrate communities collected in all sampling years was determined (Figure 21). Macroinvertebrate assemblages at upstream sites have the lowest mean temperature optima, which then increased through mid-reach and downstream sites. These differences are expected based on predictions of the River Continuum Concept (Vannote et al. 1980) and are observable in the stream itself, as colder, faster, deeper headwaters transition to shallower stretches flowing through a landscape with more human disturbance and impervious surfaces. However, within this framework, the mean community temperature optima have decreased significantly during the course of this study among downstream and mid-reach sampling sites (Figure 21A). Mean temperature optima of downstream sites show a stepwise pattern of decrease; optima in 2013-2015 are significantly lower than those of all prior years, but not significantly different from each other, suggesting that response to changed temperature regimes may have stabilized. Mid-reach communities also showed a significant decrease in mean temperature optima through 2013, and though the community means in 2014-2015 were higher than in 2013, differences were not significant, again suggesting potential stabilization. In contrast, mean temperature optima of upstream communities have changed little; the community optima in 2009 was significantly higher than 2005, but in 2011 it fell again and there are no significant differences in community temperature optima from 2011 through 2015.

Community fine sediment optima are relatively low overall and similar among all reaches, but %FSS optima in the 2005 & 2009 communities are significantly higher than the communities in 2013-2015. Among downstream and upstream sites, differences in mean %FSS optima are not significant across 2013-2015. Among mid-reach sites, the pattern of lower sediment optima seen in 2012-2014 was reversed in 2015, when mean %FSS optima increased significantly to levels similar to the earliest years of the study.

a. Macroinvertebrate community temperature optima (weighted means).



b. Macroinvertebrate community % fine sediment optima (weighted means).

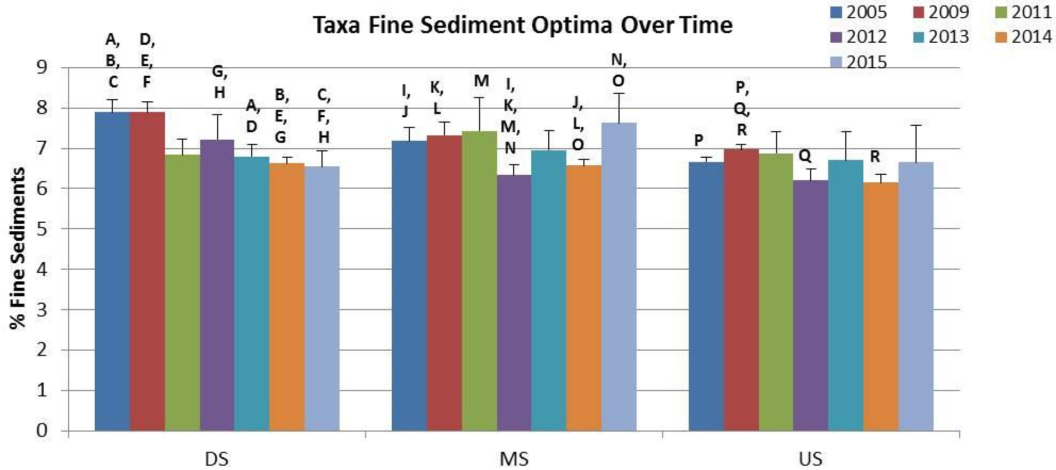


Figure 21. Temperature and % fine sediment optima among macroinvertebrate assemblages collected in Whychus Creek. Letter pairs indicate significant difference between means ($p<0.05$). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25

Functional feeding groups

Because food sources and availability play a large role in structuring aquatic macroinvertebrate communities, examining different functional feeding groups (FFGs) can be informative. The proportion of different FFGs varies naturally based on stream order, reach location, and different inputs of allochthonous (terrestrial-derived, i.e. leaves, seeds, wood, carcasses, feces) and autochthonous (stream-derived, i.e. photosynthesis by primary producers, decomposition of dead organisms, feces) nutrients into the stream (Vannote et al., 1980). However, FFG distributions are further influenced by surrounding land uses, hydrologic alterations, and excess nutrients, sediment, or contaminants.

The main FFGs of aquatic macroinvertebrates are shredders, collectors, scrapers, and predators (reviewed in Wallace & Webster, 1996). Shredders (SH) rely on terrestrial organic input such as leaf litter and wood (coarse particulate organic matter, CPOM, >1 mm diameter), and are often dominant in headwater reaches, especially where there is substantial riparian shading. Their reliance on CPOM makes them particularly sensitive to riparian conditions and surrounding land uses that impact allochthonous inputs to the stream; because SH are more specialized in their feeding and less tolerant of disturbance, their proportions are expected to be lower in more impaired habitats.

Collectors include gatherers (CG), which take in fine particulate organic matter (FPOM, <1 mm diameter) deposited on surfaces and sediment, and filterers (CF), which take in suspended particles of FPOM. Collector-gatherers are often an abundant component of the stream biota and an important prey resource for invertebrate and vertebrate predators. Because collectors are generalist feeders with a fairly broad food range, they can be more tolerant of disturbances that might alter food availability, and changes in their abundances with impairment vary.

Scrapers (SC), also called grazers, rasp algae and diatoms off mineral and organic surfaces, and can become dominant where primary production is maximized. Scrapers are a more specialized feeding group that require abundant periphyton; their proportions can be lower in impaired habitats, as the filamentous algae and mosses that increase as an early response to stream pollution interfere with their ability to feed.

Predators (PR) consume other animals by mastication or using piercing mouthparts to suck out body contents. Their proportions within the community remain relatively steady with stream order but because they rely on an abundant food base, and different life stages of a single taxon may require prey of different sizes, the PR proportions are expected to be lower at more disturbed or impaired sites.

Overall, FFG examination was not revealing. Predator taxa were present in the greatest proportions among all sampling sites and years, followed by CG, SC, and SH, with CF comprising the lowest proportion of all taxa (Fig 22). The relative diversity and abundance of both predator and shredder taxa varied greatly across years and stream reaches (Figures 23 and 25). More significant changes were seen among the more specialized SC taxa (Figure 24), with overall decreased diversity and relative abundance across time. Among collectors, CF changed more over time than CG, with increased diversity and relative abundance over time (Figure 26).

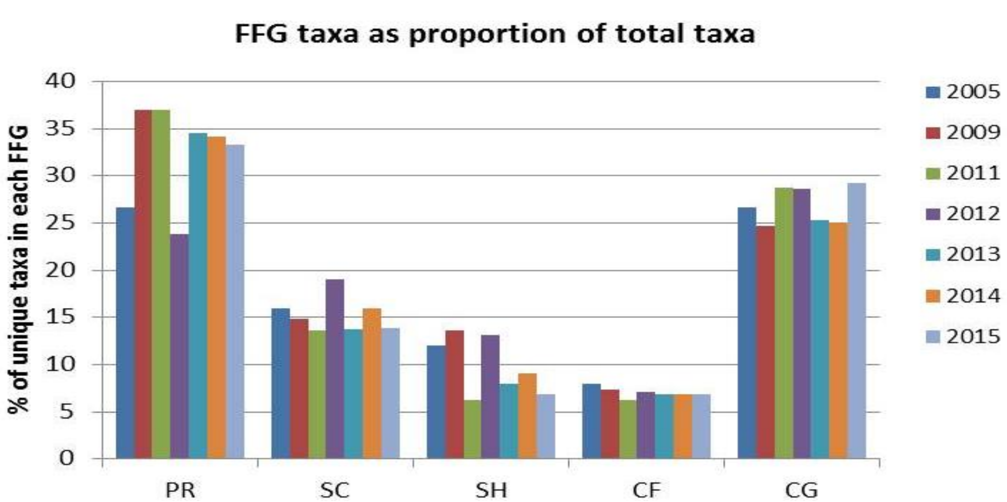
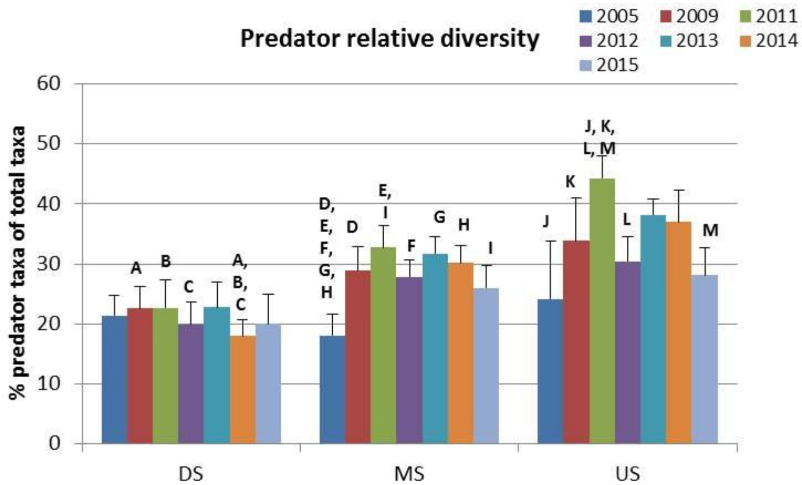


Figure 22. Proportion of total taxa collected across all Whychus Creek sampling reaches comprised by each functional feeding group. PR, predator; SC, scraper; SH, shredder; CF, collector-filterer; CG, collector-gatherer.

a. Relative diversity of predator taxa



b. Relative abundance of predator taxa

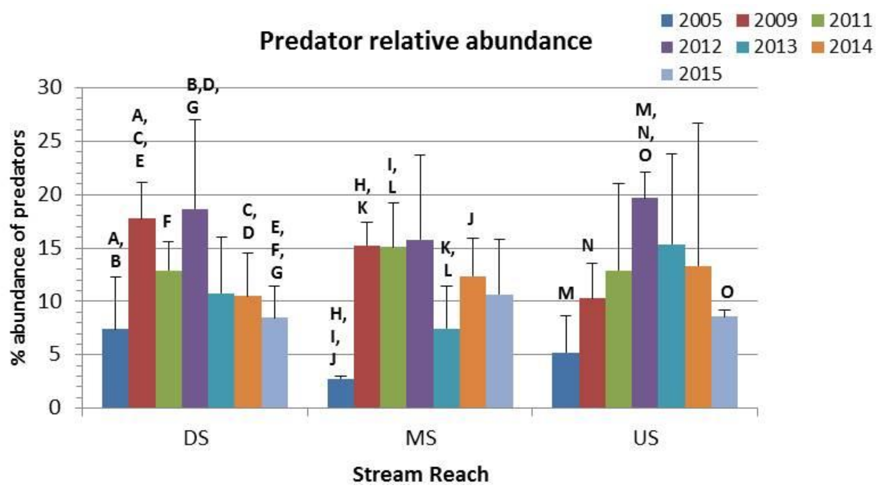
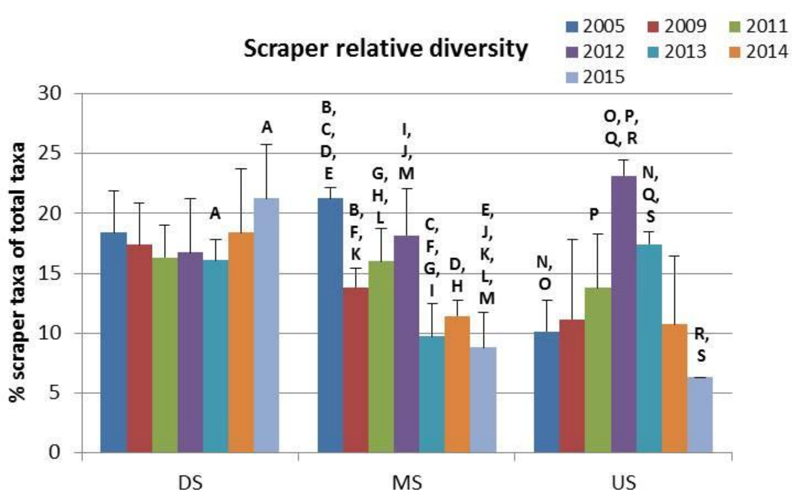


Figure 23. Changes in predator taxa over time among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (P<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

a. Relative diversity of scraper taxa



b. Relative abundance of scraper taxa

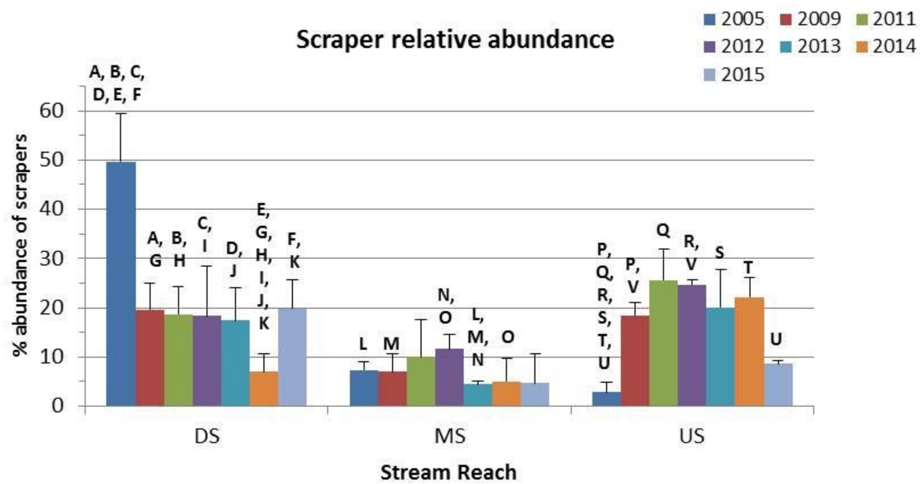
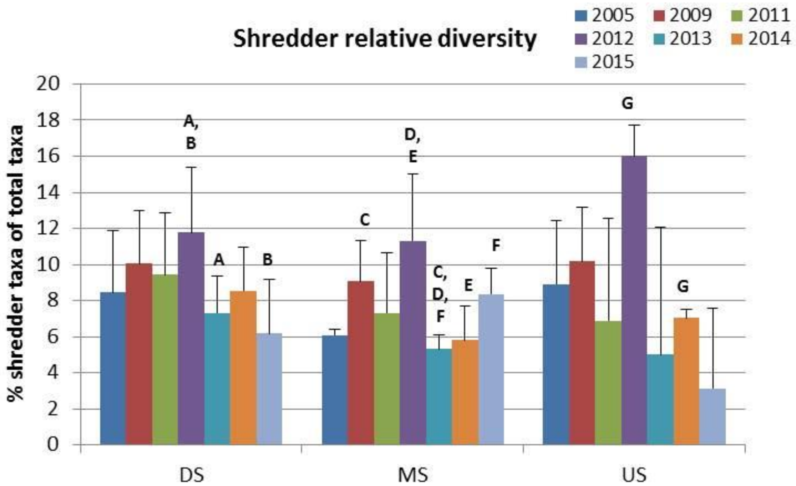


Figure 24. Changes in scraper taxa among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (p<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

a. Relative diversity of shredder taxa



b. Relative abundance of shredder taxa

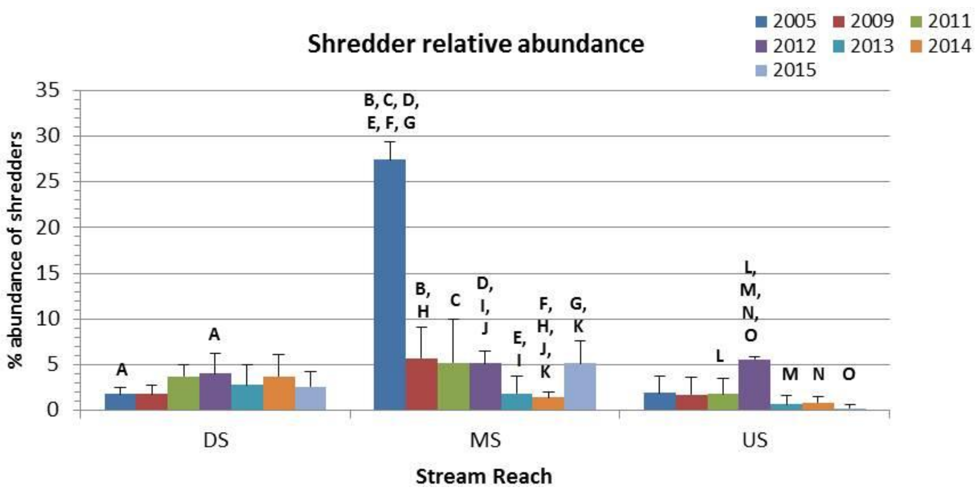
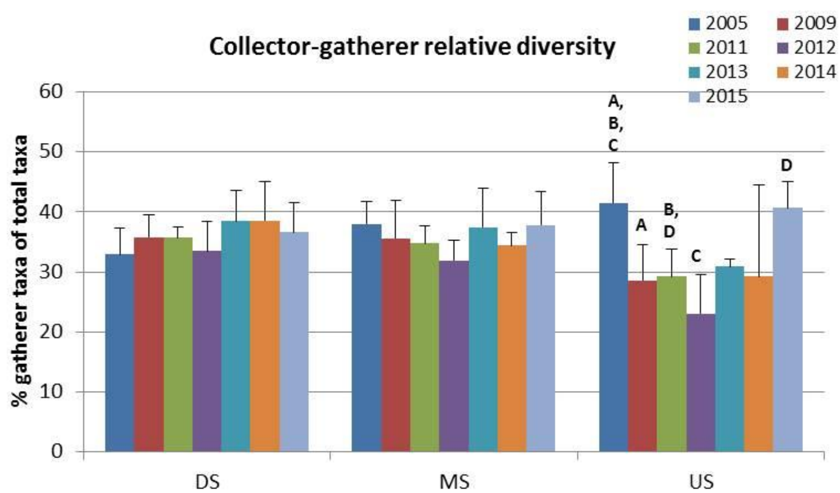
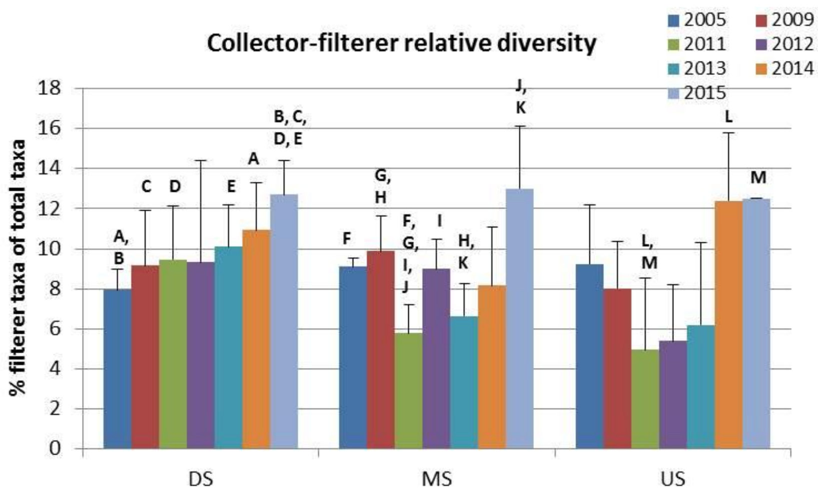


Figure 25. Changes in shredder taxa over time among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (p<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

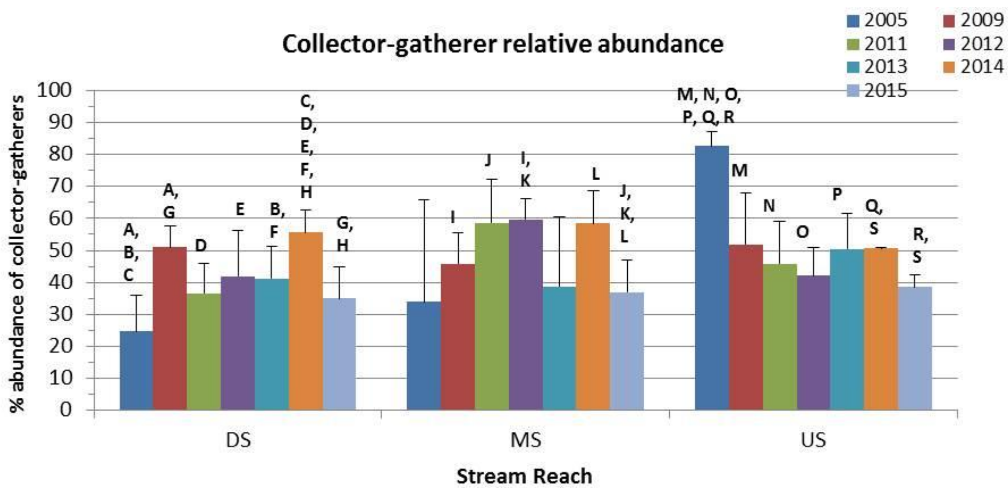
a. Relative diversity of collector-gatherer taxa



b. Relative diversity of collector-filter taxa



c. Relative abundance of collector-gatherer taxa



d. Relative abundance of collector filterer taxa

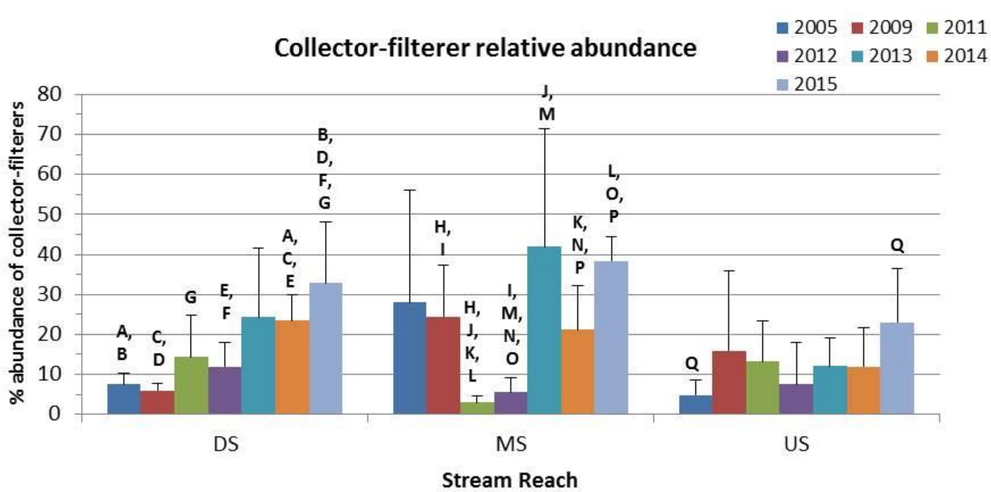


Figure 26. Changes in collector taxa (CG and CF) over time among Whychus Creek sampling sites. Letter pairs indicate significant difference between means (p<0.05). Downstream reach = RM 0.5 – 11.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Conclusions

The recovery trajectory of aquatic invertebrates in restored streams is neither linear nor instantaneous, and is influenced by a range of variables, including degree of continuing change within microhabitats in restored reaches; additional reach- or basin-scale stressors that continue to impact the restored system; distance and composition of the regional pool of colonists; interactions between species and microhabitats that 'filter' species with suitable traits from the regional species pool ; and resistance of entrenched tolerant taxa to replacement by more sensitive colonists. Thus, effectiveness monitoring of stream restoration projects must be long-term to be ecologically meaningful. In addition, studies that span only a few years may underestimate the natural variability of communities by missing disturbances that occur infrequently (i.e., major floods or fires) or as part of extended climate cycles (i.e., drought). The need for long-term biomonitoring is generally recognized (Bernhardt et al., 2005; Palmer et al., 2005), but limitations in staff, funding, and time still act to make multi-year monitoring projects the exception rather than the norm.

The Whychus Creek project study is thus noteworthy for its 10-year span; even though restoration work is still occurring, this time frame has enabled examination of long-term changes in the macroinvertebrate community within a relevant span for colonization and establishment of new taxa. It has also covered periods of different climatic conditions, with the overall lower taxa richness and organismal abundance among sampling sites likely reflecting the impacts of a dry winter and spring and an abnormally hot summer. This project has increased in complexity since its inception, when it was anticipated that biological condition scores generated by PREDATOR and IBI models would show steadily improving biological conditions resulting from stream restoration activities. These scores have been less revealing than hoped for, however; IBI scores have consistently reflected better biological conditions that PREDATOR O/E scores, but annual variations have not revealed any strong trends. PREDATOR scores have additionally changed little across the years, although O/E scores were higher among the majority of sites in 2015 compared to previous years. However, analysis of the temperature and sediment optima of the sampled community as a whole, along with that of taxa identified by PREDATOR as missing/replacement and decreaser/increasers, reveal consistently and significantly lower %FSS optima with the community over time, and among replacement and increaser taxa groups. Temperature optima within the community have also decreased significantly over time among downstream and mid-reach sampling sites.

Because similarity analyses of community assemblages showed between-year changes in community composition, additional aspects of the macroinvertebrate community have been characterized over time, many of which are more informative. Examination of individual taxonomic and ecological traits of the community suggest a measureable response to restoration activities, with increasing diversity and abundance of taxa adapted for lower temperatures and percent fine sediment loads. Community similarity analyses consistently group the 2005 and 2009 sampling communities separately from the 2011-2015 communities, indicating that earlier restoration activities had the greatest overall impact on the community. Examination of the taxa that contribute most to these community differences generally reveals groups that are more tolerant and/or associated with slower waters present in greater abundance in the earlier years, while taxa that are more abundant in later years comprise more sensitive groups (including Ephemeroptera, Plecoptera, and Trichoptera; EPT), many of which are also OR DEQ indicators for cool temperatures or low sediment conditions.

New taxa are added to the overall dataset every year, indicating that regional colonists are finding the restored reaches, and that dominance shifts may have occurred among existing resident taxa. The rate of addition of new taxa to the overall dataset has slowed with each year of sampling, but the majority of new taxa seen in each year continue to be in the sensitive EPT group.

Thus, although standard assessments such as PREDATOR and IBI have not indicated substantial improvements in biological conditions in Whychus Creek, analysis of multiple attributes and characteristics strongly suggests a community whose composition has changed in response to changing stream conditions, with a greater abundance and/or diversity of taxa that thrive in conditions with cooler temperatures and lower percent fine sediments. In recent years it seemed that the community changes were slowing and community composition stabilizing, but implementation of a new floodplain restoration project in the upstream reaches, and possibly the impacts of anomalous weather in 2015, led to decreased abundance ad taxa richness and lower values for several of the metrics used in analysis, although PREDATOR O/E scores were higher than in previous years. Increasing frequency and severity of altered precipitation patterns and wildfires may contribute to community changes in the coming years, along with other stressors likely present in the watershed that have not been remediated by recent restoration activities. Integrated analysis of macroinvertebrate community data along with water chemistry, surrounding landscape use, and climate variables in the future could provide additional insights about drivers of community changes.

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APPENDIX A Whychus Creek field data sheet

Site ID: Date: Sampled by:

Start time: _______ End time: _______ Air temp _____ °C Water temp. ______ °C

Sample Information:

# of riffles sampled: _____ Field duplicate collected: ___ yes ___ no

# of kicks composited ___ 8 x 1 ft^2 OR ___other (describe): total # field duplicate jars _____

Total # sample jars _______

Human use & influence (*check **all** that apply; right & left bank relative to observer facing downstream*)

A = absent	**B** = on bank	**C** = ≤ 30 ft from bank	**D** = > 30 ft from bank			
Disturbance	**Left bank**	**Right bank**		**Disturbance**	**Left bank**	**Right bank**
Riprap/wall/dike/armored bank				Landfill/trash		
Buildings				Park/lawn		
Industrial				Row crops		
Rural residential				Pasture/range/hay field		
Urban residential				Livestock w/stream access		
Pavement/cleared lot				Logging within last 5 years		
Road/railroad				Sand or gravel mining		
Pipes (inlet/outlet)				Forest/woodland		
Other:						

Qualitative observations:

Water odors: none **/** organic **/** rotten eggs **/** fishy **/** chlorine **/** petroleum **/** other (describe):
Water appearance: clear **/** turbid **/** milky **/** dark brown **/** foamy **/** oily sheen **/** other (describe):
Dominant land use: Forest **/** agriculture (crops / pasture) **/** urban (industrial / residential) **/** other:
Extent of algae covering submerged materials: none **/** 1-25% **/** 25-50% **/** 50-75% **/** 75-100%
Type of algae: none **/** filamentous (i.e. strands >2") **/** close-growing **/** floating clumps

Physical characteristics:
(*if your reach has fewer than 8 riffles, record characteristics only for the number of riffles sampled*)

Substrate

% composition of riffle	**Riffle1**	**Riffle2**	**Riffle3**	**Riffle4**	**Riffle5**	**Riffle6**	**Riffle7**	**Riffle8**
Bedrock (continuous rock)								
Boulder (> 12 in.; larger than basketball)								
Cobble (2.5-12 in.; tennis ball to basketball)								
Gravel (0.6-2.5 in.; marble to tennis ball)								
Sand (< 0.6 in.; smaller than marble)								
Silt/clay/muck (fine suspended particles)								
Woody debris								
Other (describe)								

Water depth
Please record units used (*check measuring tape—one side is metric, the other English*)

Parameter	**Riffle1**	**Riffle2**	**Riffle3**	**Riffle4**	**Riffle5**	**Riffle6**	**Riffle7**	**Riffle8**
Wetted width								
Depth @ ¼ wetted width								
Depth @ ½ wetted width								
Depth @ ¾ wetted width								

Additional notes or observations (including other wildlife noted):

APPENDIX B OR DEQ indicator taxa for temperature and fine sediment stressors. Values in parentheses indicate temperature (oC) or sediment (% fine sediment) optima for each taxon.

Taxon	Temperature indicator	Fine sediment indicator
Prosimulium	Cool (12.2)	---
Baetis bicaudatus	Cool (12.3)	---
Zapada columbiana	Cool (12.9)	---
Neothremma	Cool (12.9)	---
Parapsyche elsis	Cool (13.5)	Low (4)
Caudatella	Cool (13.6)	Low (4)
Megarcys	Cool (13.6)	Low (4)
Visoka	Cool (13.7)	---
Epeorus grandis	Cool (14.2)	Low (2)
Yoraperla	Cool (14.2)	---
Ephemerella	Cool (14.4)	---
Drunella coloradensis/flavilinea	Cool (14.5)	---
Doroneuria	Cool (14.5)	---
Despaxia	Cool (14.5)	---
Turbellaria	Cool (14.6)	---
Ironodes	Cool (14.9)	---
Drunella doddsi	Cool (15.2)	Low (3)
Ameletus	Cool (15.2)	---
Rhyacophila Brunnea Gr.	Cool (15.5)	Low (4)
Cinygmula	Cool (15.5)	Low (6)
Micrasema	Cool (15.6)	---
Diphetor hageni	Warm (17.9)	---
Antocha	Warm (18.3)	---
Hydropsyche	Warm (18.5)	---
Juga	Warm (18.6)	High (15)
Chironomini	Warm (18.8)	High (10)
Zaitzevia	Warm (19.0)	High (9)
Optioservus	Warm (19.6)	High (12)
Dicosmoecus gilvipes	Warm (20.6)	---
Physa	Warm (21.1)	High (21)
Arctopsyche	---	Low (2)
Rhyacophila Hyalinata Gr.	---	Low (3)
Rhyacophila Angelita Gr.	---	Low (3)
Drunella grandis	---	Low (3)
Epeorus longimanus	---	Low (4)
Rhithrogena	---	Low (5)

Rhyacophila Betteni Gr.	---	Low (5)
Glossosoma	---	Low (5)
Baetis tricaudatus	---	Low (6)
Oligochaeta	---	High (10)
Paraleptophlebia	---	High (11)
Tanypodinae	---	High (12)
Ostracoda	---	High (17)
Hydroptila	---	High (17)
Lymnaeidae	---	High (18)
Cheumatopsyche	---	High (20)
Sphaeriidae	---	High (21)
Coenagrionidae	---	High (25)

APPENDIX C Macroinvertebrate taxa list for Whychus Creek, 2005-2015

Phylum or subphylum	Class or Subclass	Order	Family	Genus	Species	2005	2009	2011	2012	2013	2014	2015
Platyhelminthes	Turbellaria					√	√	√	√	√		√
Annelida	Oligochaeta					√	√	√	√	√	√	√
Annelida	Hirudinea	Rhynchobdellida	Glossiphoniidae	Helobdella	stagnalis					√		
Nematoda						√	√	√	√	√	√	√
Arthropoda/ Crustacea	Malacostraca	Amphipoda						√		√		
Arthropoda/ Crustacea	Malacostraca	Decapoda	Astacidae	Pacifasticus			√					√
Arthropoda/ Crustacea	Copepoda							√				
Arthropoda/ Crustacea	Ostracoda					√		√	√	√		√
Arthropoda	Arachnoidea	Trombidiformes				√	√	√	√	√	√	
Arthropoda	Insecta	Coleoptera	Elmidae	Narpus		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Optioservus		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Zaitzevia		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Cleptelmis		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Ampumixis		√	√	√	√	√	√	√
Arthropoda	Insecta	Coleoptera	Elmidae	Lara	avara		√	√	√	√	√	
Arthropoda	Insecta	Coleoptera	Dytiscidae			√		√		√	√	√
Arthropoda	Insecta	Coleoptera	Dryopidae	Helichus			√					√
Arthropoda	Insecta	Coleoptera	Hydrophilidae	Ametor			√	√		√		√
Arthropoda	Insecta	Odonata	Coenagrionidae			√						

Arthropoda	Insecta	Diptera	Empididae	Neoplasta			√	√	√	√		√
Arthropoda	Insecta	Diptera	Empididae	Hemerodromia		√	√					√
Arthropoda	Insecta	Diptera	Empididae	Chelifera		√					√	√
Arthropoda	Insecta	Diptera	Empididae	Clinocera		√	√	√		√	√	√
Arthropoda	Insecta	Diptera	Empididae	Roederoides						√		
Arthropoda	Insecta	Diptera	Empididae	Wiedemannia		√						
Arthropoda	Insecta	Diptera	Tipulidae	Antocha		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Cryptolabis		√						√
Arthropoda	Insecta	Diptera	Tipulidae	Dicranota			√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Hesperoconopa		√	√				√	√
Arthropoda	Insecta	Diptera	Tipulidae	Hexatoma		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Limnophila			√	√				
Arthropoda	Insecta	Diptera	Tipulidae	Rhabdomastix			√					
Arthropoda	Insecta	Diptera	Athericidae	Atherix		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Dixidae	Dixa			√		√			
Arthropoda	Insecta	Diptera	Chironomidae	Tanypodinae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Chironominae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Diamesinae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Ceratopogoninae			√					√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Atrichopogon							√	
Arthropoda	Insecta	Diptera	Ceratopogonidae	Dasyhelea				√	√	√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Forcipomyia				√				
Arthropoda	Insecta	Diptera	Blephariceridae	Bibiocephala					√			

Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicera		√	√	√		√	√	√
Arthropoda	Insecta	Diptera	Deuterophlebiidae								√	
Arthropoda	Insecta	Diptera	Simuliidae	Prosimulium		√	√	√	√		√	
Arthropoda	Insecta	Diptera	Simuliidae	Simulium		√	√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Ephydridae			√	√					√
Arthropoda	Insecta	Diptera	Psychodidae	Pericoma		√	√			√		
Arthropoda	Insecta	Diptera	Psychodidae	Maruina			√			√		
Arthropoda	Insecta	Diptera	Tabanidae				√	√				
Arthropoda	Insecta	Diptera	Sciomyzidae							√		
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella		√	√	√	√	√		
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella	turbida		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella	insignificans							√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis		√	√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis	tricaudatus		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Diphetor	hageni	√	√		√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Labiobaetis								√
Arthropoda	Insecta	Ephemeroptera	Ameletidae	Ameletus		√	√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella		√	√		√			√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella	delantala							√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella	margarita			√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Serratella		√						
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella		√	√	√	√			√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella (Serratella)	tibialis		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	dorothea			√				

Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	excrucians		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	edmundsi						√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	hystrix	√	√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	spinifera	√		√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	flavilinea				√			
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	doddsi					√	√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	coloradensis		√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus		√	√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	albertae			√	√	√		√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	deceptivus/ hesperus			√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	grandis		√	√	√		√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	longimanus		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Rhithrogena		√	√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygmula		√	√	√	√	√		√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygma					√			
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	Tricorythodes		√						
Arthropoda	Insecta	Ephemeroptera	Leptophlebiidae	Paraleptophlebia		√	√	√	√	√	√	√
Arthropoda	Insecta	Hemiptera	Corixidae								√	
Arthropoda	Insecta	Hemiptera	Gerridae								√	
Arthropoda	Insecta	Megaloptera	Sialidae	Sialis		√						
Arthropoda	Insecta	Plecoptera	Perlidae			√	√	√				
Arthropoda	Insecta	Plecoptera	Perlidae	Calineuria	californica	√	√					
Arthropoda	Insecta	Plecoptera	Perlidae	Hesperoperla		√						
Arthropoda	Insecta	Plecoptera	Perlidae	Doroneuria						√		

Arthropoda	Insecta	Plecoptera	Perlodidae			√	√	√	√	√	√	
Arthropoda	Insecta	Plecoptera	Perlodidae	Isoperla			√	√			√	
Arthropoda	Insecta	Plecoptera	Perlodidae	Megarcys			√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Rickera	sorpta		√	√			√	
Arthropoda	Insecta	Plecoptera	Perlodidae	Kogotus/Rickera		√				√	√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Skwala		√	√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Paraperla		√			√		√	
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Suwallia			√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Sweltsa		√	√	√		√	√	√
Arthropoda	Insecta	Plecoptera	Leuctridae				√	√				
Arthropoda	Insecta	Plecoptera	Leuctridae	Despaxia	augusta		√					
Arthropoda	Insecta	Plecoptera	Nemouridae	Amphinemura					√			
Arthropoda	Insecta	Plecoptera	Nemouridae	Visoka	cataractae	√	√	√	√		√	√
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada		√	√	√	√			
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	cinctipes		√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	columbiana		√		√	√		
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	oregonensis			√			√	
Arthropoda	Insecta	Plecoptera	Pteronarcyidae	Pteronarcys		√	√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Yoraperla		√						
Arthropoda	Insecta	Plecoptera	Capniidae			√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Apataniidae	Pedomoecus			√			√		
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Agapetus		√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Glossosoma		√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Arctopsyche	grandis	√	√					√
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsyche		√	√	√	√	√	√	√

Arthropoda	Insecta	Trichoptera	Hydropsychidae	Parapsyche	elsis		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Brachycentridae	Micrasema		√	√	√	√		√	√
Arthropoda	Insecta	Trichoptera	Brachycentridae	Brachycentrus	americanus	√	√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Helicopsychidae	Helicopsyche		√						
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila		√	√	√	√	√	√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Angelita Gr.			√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	arnaudi		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	atrata complex					√		
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Betteni Gr.		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Brunnea/ Vemna Gr.		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	coloradensis						√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Hyalinata Gr.		√	√			√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	narvae		√	√				
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Nevadensis Gr.			√				
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	grandis		√					
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Vagrita Gr.		√	√			√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	valuma		√	√	√			
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	vetina complex					√		
Arthropoda	Insecta	Trichoptera	Sericostomatidae	Gumaga					√			
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Agraylea		√						
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptila		√	√	√		√	√	√
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Metrichia		√						

Arthropoda	Insecta	Trichoptera	Hydroptilidae	Ochrotrichia		√	√			√		√
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Stactobiella				√	√			
Arthropoda	Insecta	Trichoptera	Lepidostomatidae	Lepidostoma			√			√	√	
Arthropoda	Insecta	Trichoptera	Philopotamidae	Wormaldia		√			√	√		
Arthropoda	Insecta	Trichoptera	Philopotamidae	Dolophilodes		√	√	√				
Arthropoda	Insecta	Trichoptera	Limnephilidae			√		√	√			
Arthropoda	Insecta	Trichoptera	Limnephilidae	Dicosmoecus		√						√
Arthropoda	Insecta	Trichoptera	Limnephilidae	Onocosmoecus		√						
Arthropoda	Insecta	Trichoptera	Limnephilidae	Psychoglypha		√			√	√		
Mollusca	Gastropoda	Basommatophora	Ancylidae	Ferrissia		√					√	√
Mollusca	Gastropoda	Basommatophora	Physidae	Physa		√			√		√	
Mollusca	Gastropoda	Neotaenioglossa	Pleuroceridae	Juga			√		√	√	√	
Mollusca	Gastropoda	Basommatophora	Planorbidae			√			√			
Mollusca	Bivalvia	Pelecypoda	Pisidiidae							√	√	√

Native Fish Monitoring in Whychus Creek

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Abstract

Restoration actions on Whychus Creek aim to restore the stream habitat, flows, fish passage, and water quality necessary to support self-sustaining populations of reintroduced Chinook salmon (*Oncorhyncus tshawytscha*), anadromous steelhead and resident redband trout (*Oncorhyncus mykiss*), and bull trout. Hundreds of thousands of steelhead and Chinook fry and thousands of smolts have been released annually since 2007 and 2009, respectively. The Whychus Creek Monitoring Plan identified fish populations monitored annually by Portland General Electric as a biological indicator of restoration effectiveness. But, fish population data are confounded by the ongoing release of hundreds of thousands of juvenile steelhead and salmon annually in concert with implementation or recent completion of multiple restoration projects along the creek, limiting the applicability of these data to evaluate restoration effectiveness. While recognizing this limitation, the Upper Deschutes Watershed Council summarizes PGE's and other available fish monitoring data for Whychus Creek annually to track the status and trends of fish populations. Although *O. mykiss* have accounted for the majority of fish caught in Whychus in every year since 2007, densities remain low with no significant increasing or decreasing trend. A 2015 genetic analysis of 2005 and 2013 Whychus *O. mykiss* populations showed the 2013 population dominated (75%) by Round Butte Hatchery stock, suggesting steelhead fry released in Whychus are residualizing in the creek with potential impacts to the natural-origin O. mykiss population. No Chinook smolts were released and therefore none captured in 2015; Chinook densities in previous years were similarly low as juvenile steelhead densities. Redd densities averaged 6.5 redds/km in 2015 in mainstem sites, slightly down from the 2014 estimate of 9.5 redds/km. An estimated 6,223 (± 1,797) naturally reared steelhead smolts migrated out of Whychus in 2015 between March and June, with migration peaking in May. Adult migration data show steelhead migration into the Upper Deschutes occurring from September to May and Chinook from May through July. Seven returning adult Chinook and four returning adult steelhead ascended the Deschutes River in 2015, one of each of which ascended Whychus Creek. Continued monitoring and improved information about Chinook, steelhead, and redband life histories and habitat use in Whychus Creek will help restoration partners refine strategies to optimize conditions for these species.

Introduction

Anadromous populations of summer steelhead (*Oncorhyncus mykiss*) and spring Chinook salmon (*Oncorhyncus tshawytscha*) were extirpated from the Upper Deschutes subbasin following completion of the Pelton-Round Butte hydroelectric project dams in 1964. With dam re-licensing in 2005, Portland General Electric and the Confederated Tribes of Warm Springs agreed to restore anadromous populations in the Upper Deschutes subbasin. Steelhead fry were reintroduced in Whychus Creek and the Crooked River system in 2007 and have been released in the hundreds of thousands every year since; Chinook fry and smolts, and steelhead smolts, were first released in 2009 (Table 2). Under the 2005 FERC re-licensing agreement for the Pelton-Round Butte hydroelectric project, Portland General Electric (PGE) conducts native fish monitoring annually in Upper Deschutes sub-basin rivers and tributaries supporting salmon and steelhead reintroduction, and publishes multiple reports summarizing monitoring results. Objectives of PGE's native fish monitoring include describing juvenile *O. mykiss* and Chinook salmon population size and size-frequency distributions, adult spawning activity, and timing of spawning. PGE additionally monitors juvenile migration from Upper Deschutes tributaries into Lake Billy Chinook to estimate smolt production and establish migration timing and rate, counts adult returns to the Pelton Fish Trap, and tracks adult migration above Round Butte dam.

Table 2. Steelhead and Chinook fry and smolts stocked in Whychus Creek by year.

	Steelhead		Chinook	
Year	Fry	Smolts	Fry	Smolts
2007	275,000	-	-	-
2008	290,650	-	-	-
2009	278,823	5,000	71,603	5,000
2010	229,797	3,600	73,613	5,207
2011	288,768	5,456	72,898	6,504
2012	248,131	4,871	53,647	6,898
2013	291,921	2,209	87,896	5,005
2014	310,900	4,498	0	0
2015	216,000	25,428	0	15,576

Restoration partners on Whychus Creek aim to restore the stream habitat, flows, passage and water quality necessary to support appropriate life history stages of steelhead and Chinook, as well as resident redband and bull trout (UDWC 2009). A key component of the restoration strategy is long-term monitoring to 1) track the status and trends of selected biological and physical indicators of stream conditions, and 2) evaluate the effectiveness of restoration actions. Fish populations were identified as a biological indicator under the original monitoring plan (UDWC 2009) for two reasons: PGE native fish monitoring data would be available under PGE monitoring agreements for years into the future, and anadromous fish populations were a target conservation value of the restoration effort.

Kunkel (2010) evaluated the relative utility of fish populations in Whychus Creek as biological indicators of restoration effectiveness. His report identified multiple obstacles to the use of available data to evaluate the short-term response of Whychus fish populations to changes in habitat resulting from restoration actions. Foremost among these are annual releases of steelhead fry and smolts, which at least in part drive *O. mykiss* abundance and mask any response of the natural-origin population to changes in stream conditions. Releases are scheduled to continue until it is determined that steelhead populations may be sufficiently established as to be self-sustaining (ODFW and CTWS 2008). Attempts to estimate abundance of juvenile steelhead versus redband trout are confounded by the inability to differentiate juvenile steelhead (hatchery outplants) and redband (natural-origin) without conducting expensive genetic analyses. These efforts may be further complicated by alterations to the genetic structure of Whychus *O. mykiss* that have occurred since reintroduction began. With few exceptions, short term changes to habitat following restoration do not reflect the fully restored condition of the creek and may even adversely affect fish populations; independent of population fluctuations resulting from ongoing yearly releases, fish population trends may not begin to reflect habitat suitability until years after restoration projects are completed, once sediments, stream banks, and aquatic and riparian vegetation conditions have stabilized.

Given these obstacles, available fish population data is of limited use over the short term as a biological indicator of restoration effectiveness in Whychus Creek. We anticipate native fish monitoring data may become a useful indicator of restoration effectiveness over the long term, as restored communities achieve full ecological function. A USFWS and USFS genetic analysis completed in 2015 showed the 2013 juvenile O. mykiss population was dominated (75%) by Round Butte Hatchery stock, suggesting fry reintroduced into Whychus are residualizing and displacing natural origin fish (Adams *et al* 2015); PGE will conduct additional genetic analysis to determine relative proportions of juvenile redband (natural origin) and steelhead (hatchery origin) in 2017 and 2022, five and ten years after returning fish are first passed upstream of the hydroelectric project. These data will provide further insight into population dynamics and interactions between the two life histories, and between Round Butte and natural-origin stock. In the interim, UDWC continues to track PGE's native fish monitoring on Whychus Creek and summarize their findings in an annual technical report. As restoration projects are completed and additional fish population data become available, UDWC will continue to evaluate the use of these data as a biological indicator of restoration effectiveness.

We compiled data specific to Whychus Creek and the Deschutes River from PGE's 2016 Fisheries Monitoring reports, including: 1) juvenile *O. mykiss* and Chinook salmon rearing densities (Madden and Bennett 2016); 2) *O. mykiss* redd densities (Madden and Bennett 2016); 3) steelhead and Chinook salmon smolt outmigration (Hill and Quesada 2016); and 4) adult steelhead and Chinook returns and migration (Burchell et al 2016). We compare 2016 data to 2007-2015 results; 2006 native fish monitoring data were collected using different methods and are not comparable to 2007-2015 data, and are therefore not considered in this report. We also note USFS and ODFW fish surveys conducted in 2015, but do not report findings as data from these surveys were not yet available at the time of writing of this report.

Fish populations in Whychus Creek

Historically, Whychus Creek provided important spawning and rearing habitat for anadromous summer steelhead (*Oncorynchus mykiss*), Chinook salmon (*Oncorhynchus tshawytscha*) and pacific lamprey (*Lampetra tridentata*). Whychus Creek constituted the primary steelhead spawning area in the Deschutes basin. The construction of the Pelton Round Butte hydroelectric dams led to the extirpation of anadromous fish species from the upper Deschutes River and its tributaries during the 1960s. The dams fragmented the remaining populations of resident fish species by preventing migration between the lower and upper Deschutes sub-basins.

Fish species presently occurring in Whychus Creek include resident redband trout and reintroduced steelhead trout (*Oncorynchus mykiss*), reintroduced Chinook salmon (*Oncorhynchus tshawytscha*), bull trout (*Salvelinus confluentus*), non-native brown trout (*Salmo trutta*), longnose dace (*Rhinichthys cataractae*), and sculpin (*Cottidae*). Non-native brook trout (*Salvelinus fontinalis*) were caught during native fish monitoring surveys in 2007 and 2008 but have not been observed since; bridgelip sucker (*Catostomus columbianus*) were last observed in 2006, also during PGE's native fish monitoring. No current sampling effort specifically targets either of these species, but they are believed to persist at low abundance in Whychus Creek (M. Hill 2011, personal communication). Native bull trout (*Salvelinus* *confluentus*) have been observed in Whychus Creek below Alder Springs (Fies *et al* 1996). PGE captured one bull trout each year in the Alder Springs area from 2003-2005 (M. Hill 2009, personal communication); two bull trout were found in Whychus in 2014, one at Alder Springs (Madden *et al* 2015) and one approximately eight miles upstream at Rimrock Ranch (E. Porter 2015, personal communication).

Chinook salmon

Chinook use of Whychus Creek in the early 1950's appears to have been consistent although low, with spawners and redds numbering from single digits to the low teens, and limited to the lower few miles of the creek (Nehlsen 1995). Chinook spawning in Whychus diminished through the late '50s, with the last spawners and redds counted in 1959. Chinook reintroduction efforts are focused on Whychus Creek and the Metolius River sub-basin (ODFW and CTWS 2008). The preliminary escapement goal for upper basin spring Chinook salmon is 1000 adults annually above PRB; a model simulation for Chinook recovery in the Metolius Basin (not including Whychus) estimates annual smolt production of approximately 350 smolts through 2040 (ODFW and CTWS 2008).

Sockeye salmon

Sockeye salmon (*Oncorhyncus nerka*) historically occurred in Suttle Lake (Metolius sub-basin), but they probably did not occur in Whychus Creek due to the lack of access to a lake system necessary for juvenile sockeye rearing. Kokanee salmon, the landlocked form of sockeye, now utilize Lake Billy Chinook for rearing. These kokanee may be descended from Suttle Lake sockeye that were trapped behind the dams. Fies *et al* (1996) reported an observation of 11 kokanee salmon adults (spawners) in Whychus Creek downstream from Alder Springs during a survey in 1991. This may indicate a potential for anadromous sockeye salmon to spawn in Whychus Creek and rear in Lake Billy Chinook if runs are reestablished above the dams.

O. mykiss

Redband trout and summer steelhead trout are both classified as *Oncorynchus mykiss* (Behnke 2002). Redband exhibit a resident life history behavior and spend their entire life within a stream system, although they may migrate within the system. Small numbers of redband trout in the upper Deschutes River system have historically migrated between Lake Billy Chinook and tributary streams (Groves *et al* 1999). Summer steelhead are anadromous, with juveniles rearing in streams for 1-3 years, migrating to the ocean where they remain for 1-3 years, then returning to their natal watersheds as adults to spawn. Adult steelhead may survive after spawning, return to the ocean, and then return again to streams to spawn, although Behnke (2002) reports the rate of repeat spawning of steelhead to generally be less than 10% in most populations.

Steelhead adults and redds numbered in the low hundreds in Whychus Creek throughout the 1950s but declined precipitously with the construction of the Pelton and Round Butte dams, and were eliminated altogether when fish passage efforts were abandoned (Nehlsen 1995). The reintroduction plan identifies a preliminary escapement goal of 955 adult summer steelhead. A simplistic model simulation estimates smolt production for Whychus Creek at 450 smolts through 2040 (ODFW and CTWS 2008).

Redband trout and summer steelhead naturally coexist in the lower Deschutes River downstream from the Pelton Round Butte dams. Resident and anadromous forms of *O. mykiss* may have both historically occurred in Whychus Creek as well. The 2007 release of the first reintroduced steelhead fry into

Whychus Creek was accompanied by much uncertainty about the extent to which both life history forms will again coexist in Whychus Creek. Various studies have shown both reproductive isolation and inter-breeding between resident redband and anadromous steelhead life histories. Zimmerman and Reeves (1999) provide evidence that steelhead and redband trout in the lower Deschutes River are reproductively isolated by their utilization of different spawning habitats and by differences in their time of spawning. Behnke (2002) also suggests that populations of resident and anadromous forms of *O. mykiss* may maintain their genetic distinction by spawning in separate areas within the same stream system. Conversely, a recent study from the Hood River showed that up to 40% of anadromous steelhead genes in a given generation were from wild redband trout, suggesting extensive inter-breeding between the two life histories (Christie *et al* 2011). Ackerman *et al* (2007) and Cramer and Beamesderfer (2006) had suggested that Whychus Creek will produce primarily anadromous, not resident, *O. mykiss*, based on stream flows and temperature.

A 2015 USFWS and USFS genetic analysis (Adams *et al* 2015) showed that eight years after the initial reintroduction of juvenile steelhead into Whychus Creek, 75% of mature O. mykiss in the creek assigned to Round Butte hatchery stock, indicating reintroduced steelhead fry are residualizing in the creek and displacing natural-origin redband. The majority of *O. mykiss* included in the study clearly (86-100% probability) assigned to either Round Butte hatchery or to natural-origin Whychus stock, showing the high proportion of Round Butte hatchery stock is a result of released fish residualizing rather than a result of interbreeding between reintroduced steelhead and natural-origin Whychus redband. Data are not yet available to evaluate potential spawning interactions between hatchery and natural-origin O. mykiss.

Methods

O. mykiss and spring Chinook juvenile density

PGE fisheries managers selected four study reaches in 2002 (Figure 1) representative of the range of habitats in Whychus Creek (Lewis 2003). A fifth reach was added in 2009. Naming conventions for the reaches have varied over the years; we refer to reaches by name rather than by reach numbers for clarity. The Alder Springs survey site is located downstream from Alder Springs at river mile (rm) 1.5/river kilometer (rkm) 2.5. Road Crossing is downstream from USFS Road 6360 at rm 6 (rkm 9). Camp Polk (rm 19/rkm 25.5) was sampled from 2006 through 2011 but sampling at this site was discontinued in 2012 following diversion of the stream from the straightened channel, where prior sampling had occurred, into the restored meadow channel. The Sisters site is downstream from Hwy 20 in Sisters at rm 23.5 (rkm 34.5). Wolftree-Aspen Hall is located at rm 17.5 (rkm 25).

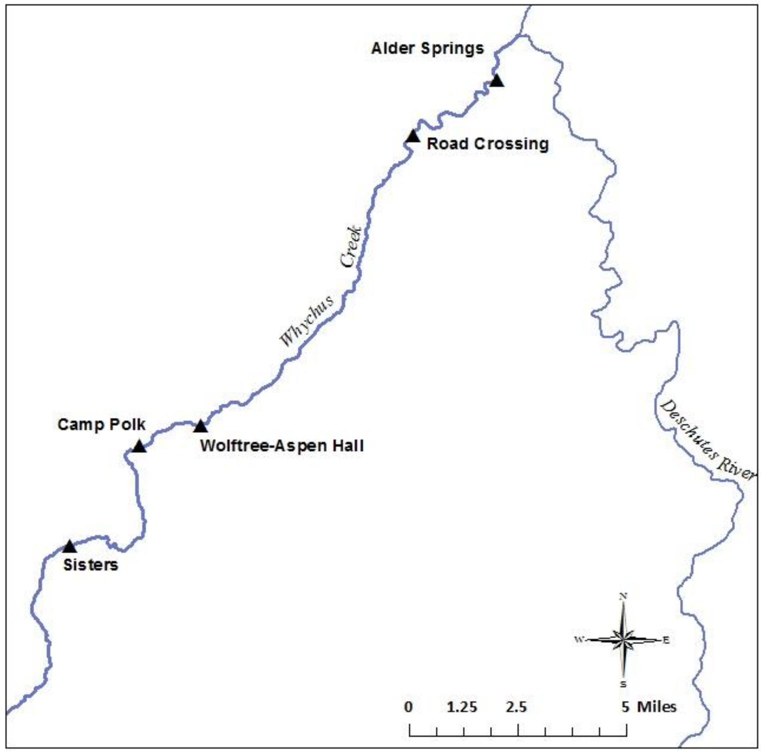

Figure 1.
Study reaches on Whychus Creek for fish population estimates. Alder Springs, Road Crossing and Sisters sites have been surveyed annually since 2006; Camp Polk was surveyed by PGE from 2006-2011; and Wolftree-Aspen Hall was surveyed annually from 2009 through 2012 and in 2014 and 2015, but was not surveyed in 2013 due to high and turbid flows. Adapted from Madden *et al* 2016.

PGE conducted fish population sampling during the low flow period, from September 22 to October 1, 2015. Study reach lengths ranged from 105-227 m, determined by the location of habitat characteristics allowing the secure placement of block nets. Block nets were situated above and below survey sections within each reach, with an additional net placed midsection to evaluate block net effectiveness and the mark-recapture sampling assumption of a closed population.

Where high flows or aquatic vegetation prevented effective use of block nets, researchers used natural habitat breaks thought to constrain fish movement, such as riffle/pool breaks, to define the beginning and end of reaches. For reaches where block nets weren't used, mark-recapture sampling was conducted from the stream section below the reach to the stream section above to determine whether fish were moving out of the reach (testing the assumption of a closed population). Where any marked fish were recaptured in the stream section below or above the reach, all *O. mykiss* captured in these sections were included in the population estimate for that reach. Where no marked fish were recaptured in these sections, fish captured in the same sections were not included in the population estimate for that reach, although they were still used for length frequency data analysis.

Mark-recapture electrofishing was conducted following protocols adapted from ODFW (Scheerer *et al* 2007), as described in Madden *et al* (2016). All fish captured were recorded by species. *O. mykiss* > 60 mm were anaesthetized, measured and marked. Fish population estimates were calculated using Chapman's modification of the Peterson mark recapture formula to reduce overestimates of population size. Length frequency distributions were compared for years before and after steelhead reintroduction.

From 2009 through 2011 and in 2013 PGE conducted snorkel surveys at three sites in Whychus Creek (6360 Road Crossing, Wolftree, and at Sisters) to generate density estimates for juvenile Chinook. The Sisters site was not snorkeled in 2011 because no Chinook fry were released into this reach in 2011. Juvenile Chinook salmon snorkel surveys were discontinued in Whychus Creek in 2012 in favor of mark-recapture electrofishing, which has proven a more effective method for sampling juvenile Chinook in Whychus. No Chinook fry were released into Whychus in 2014 due to a shortage of Chinook fry from Round Butte Hatchery, and no Chinook fry were released in any reintroduction river or tributary in 2015 due to disease-related mortality of broodstock at the hatchery (T. Shrader, personal communication, September 30, 2016).

In September 2009 USFS sampled O. mykiss in a 264-m reach downstream of the Three Sisters Irrigation Diversion to generate a pre-restoration population estimate for the project (Reischauer and Dachtler 2009). They followed the same protocol used by PGE (Quesada and Hill 2009). USFS and ODFW sampled reach 4 of the Whychus Canyon restoration project in 2015, prior to project implementation in 2016.

O. mykiss and spring Chinook smolt production

To estimate numbers of steelhead, Chinook and redband juveniles outmigrating from Whychus Creek and other tributaries and to mark downstream migrants for later identification, PGE has operated screwtraps annually on the Metolius, Deschutes and Crooked Rivers, their tributaries, and/or respective arms of Lake Billy Chinook. Recovery of marked fish provides data to estimate the relative contribution of smolts from each tributary, and, if measures are taken in the future to differentiate redband and steelhead, will provide information on out-migration of redband trout. Fish traps deployed in Whychus Creek in 2009 and 2010 were difficult to operate effectively due to vandalism and widely fluctuating flows during the downstream migration period (Hill and Quesada 2010, Hill and Quesada 2011). Resulting 2009 and 2010 data were inadequate to develop smolt production estimates. In 2011 a screwtrap deployed on the Upper Deschutes River Arm of Lake Billy Chinook replaced traps deployed in Whychus; in 2012 and 2013, the screwtrap was replaced with a Merwin trap in the same location (Figure 2). PGE sought but did not receive a permit to deploy a screwtrap at FS-Road 6360 at rm 6 on Whychus Creek in 2014.

In 2015 PGE operated a screwtrap at FS-Road 6360 from March 8 until June 1, 2015, when smolt catch fell below ten smolts per week, according to methods detailed in Hill and Quesada 2016. The trap was checked 7 d/week over this duration, with the exception of during outages on March 9 and 24, and from April 4-13, resulting from low flow events. Naturally-reared steelhead and Chinook smolts were measured, checked for a PIT tag, and tagged if they met size criteria and did not have a tag. Tagged fish were released into Whychus Creek upstream of the trap to allow calculation of trap efficiency.

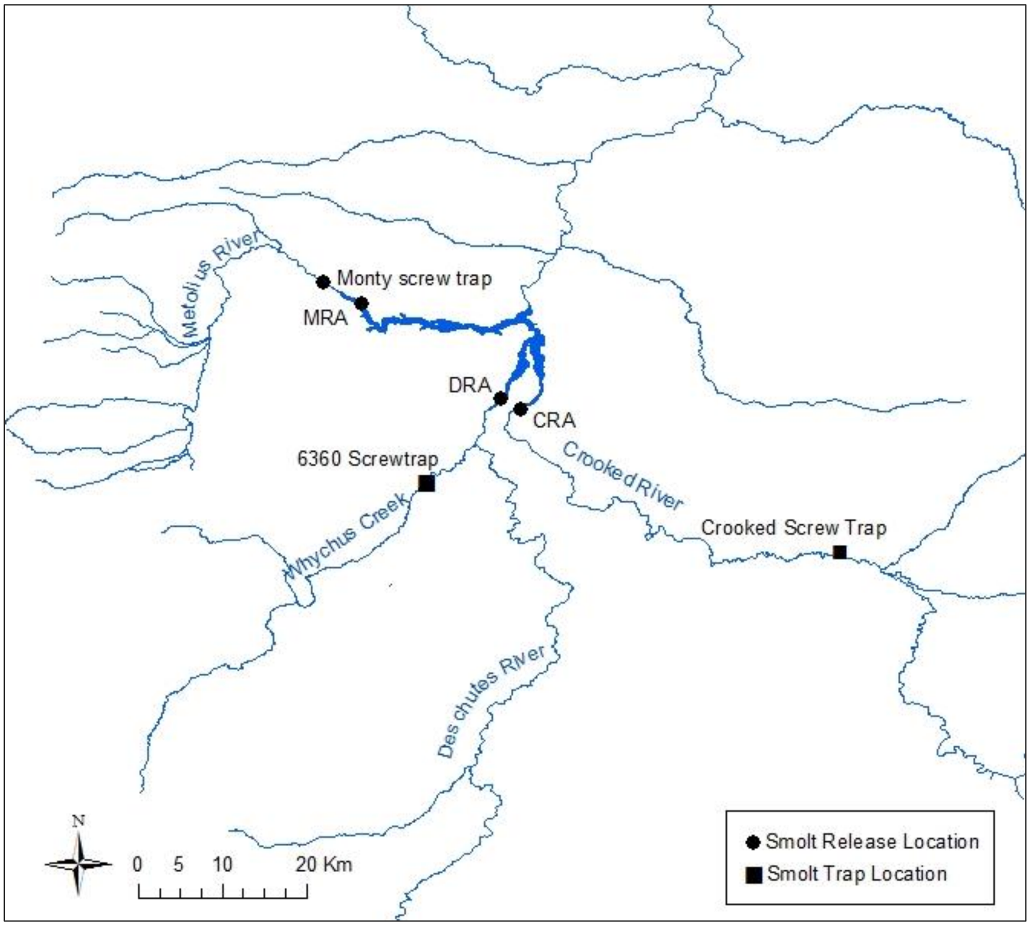

Figure 2.
2015 trap locations. Reproduced with permission from Hill and Quesada 2016.

O. mykiss redd surveys

In 2006, PGE identified four sites along Whychus Creek for *O. mykiss* redd surveys. The four sites were subdivided into ten individual index reaches to help identify the distribution of redds. PGE and Deschutes National Forest surveyed these four sites every two weeks from March through July from 2006 – 2009.

In 2010 PGE revised redd count methods to implement a spatially balanced rotating panel design recommended by American Fisheries Society Protocol. This design, similar to that used by the ODFW Coastal Salmonid Inventory Project (ODFW 2007), allows estimates of redds per kilometer and spawning distribution, reduces bias, and addresses shortcomings of the original index reach design, specifically, that index reaches may not be a reliable means to determine trends in spawning abundance because spawning site selection may not be consistent or predictable between years (Isaak and Thurow 2006). The rotating panel design incorporates two annually sampled index sites with two sites randomly selected from a predetermined set of reaches and sampled at regular, less frequent intervals (Gallagher et al. 2007).

Since 2010, PGE has sampled the two 1-km index sites, two 1-km randomly selected rotating panel sites, and three of the original 2006-2009 reaches retained to establish a population trend, identify the temporal and spatial *O. mykiss* spawning distribution, and evaluate fish use of the new channel at Camp Polk (RKM 25) (Table 2). From 2010-2012 PGE sampled one original 2006-2009 reach additional to the three sampled continuously since 2006; in 2013 and 2014 PGE randomly selected a single rotating panel reach. In 2015 Alder Springs Creek and Lewis Woodpecker Creek were re-classified as tributaries and redds detected in these reaches were not included in calculation of redds per kilometer.

To establish redband spawning timing (temporal distribution), surveyors counted redds every two weeks from March through July. One or two surveyors walked downstream at each site to identify redds and placed flagging next to each redd detected to avoid recounting redds on subsequent surveys. Surveyors also collected temperature data.

Table 2. Whychus Creek O. mykiss redd sampling sites

Site Description	Site Type
RKM 2	Index
RKM 27	Index
Lewis Woodpecker Creek	Trend (Original)
Alder Springs Creek	Trend (Original)
RKM 25	Trend (Original)
Upstream of Alder Springs Creek	Original
Rimrock Ranch/RKM 14 (2012)	Random (2012) (Original)
RKM 26	Original
Upstream of Road 4606 Footbridge	Original
RKM 38	Random (2012) (Original)
RKM 8	Random (2010)
RKM 11	Random (2011)
RKM 21	Random (2013, 2014)
RKM 22	Random (2015)
RKM 23	Random (2010)
RKM 34	Random (2011)
RKM 36	Random (2015)

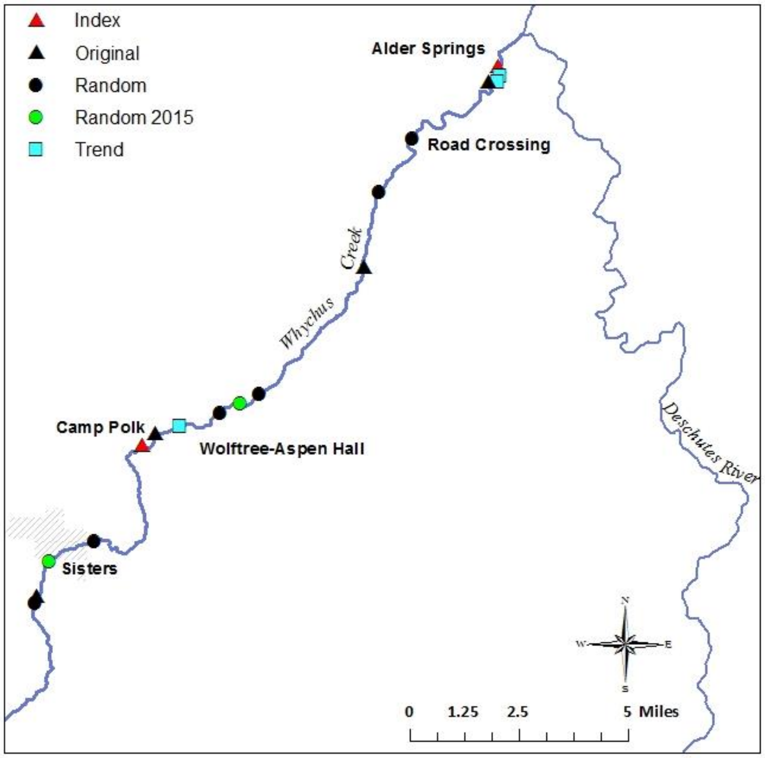

Figure 3.
Redband redds were counted at seven sites in 2015: two index sites (red triangles); two random sites (green circles); and three trend sites. The index sites and trend sites were surveyed from 2006-2015.

Adult returns and migration

Numbers of adult steelhead and Chinook returning to the Pelton trap were recorded by date. In 2015, all returning adult steelhead and Chinook salmon of known origin (identified by an intact adipose fin and a left or right maxillary clip) that returned to the Pelton Trap were tagged with two fluorescent green anchor tags, to distinguish them from conspecifics on spawning grounds, and passed upstream of Round Butte dam. Adults meeting criteria for condition, minimum size, maxillary clip and presence of a PIT tag were tagged with Juvenile Combined Acoustic Radio Telemetry (JCART) tags that emit both a radio and an acoustic signal to allow PGE biologists to track fish using either method.

Fish were tracked from fixed radio or acoustic telemetry stations at the upper end of the Deschutes, Crooked, and Metolius river arms of Lake Billy Chinook, as well as from three additional acoustic stations in the Metolius river arm. Fixed stations were programmed to run 24 hours/day, seven days/week, and recorded date and time of detection, signal strength and direction. Once a JCART-tagged fish was detected in a tributary, mobile tracking ensued. Surveyors recorded locations using GPS and field maps, and recorded spawning activity in the surrounding area including whether fish were on active redds.

Results

Species composition

As in previous years, in 2015 the majority of fish captured in Whychus Creek were *O. mykiss* including both resident redband and released steelhead. Other species captured included brown trout, speckled dace, and sculpin. One 88 mm bull trout was captured in the Alder Springs reach (Reach 1) during 2014 sampling, potentially indicating active bull trout spawning and rearing in this reach. No bull trout were captured during PGE electrofishing surveys in 2015.

O. mykiss population estimates

Whychus Creek *O. mykiss* population estimates continue to vary widely between years and among reaches, with no statistically significant increasing or decreasing trend detectable over eight years of sampling (Table 3, Figure 4). Estimated densities for any reach have rarely exceeded 40 *O. mykiss*/100m^2, compared to densities in McKay and Ochoco Creeks ranging from the 40s up to 281 *O. mykiss*/100m^2 at one site in 2014. Density estimates for 2015 remained relatively consistent with those for 2014, with somewhat higher estimates for Alder Springs and Sisters and a lower estimate for Wolftree than in 2014 and previous years. Size distribution of *O. mykiss* from 2007-2015, since steelhead reintroduction, has included a greater proportion of captured fish less than 100 mm in length than from 2002-2006, prior to steelhead reintroduction, suggesting that releases of steelhead fry have increased the relative proportion of fish in this size class. USFS and ODFW 2015 data were not available at the time of writing this report.

Table 3. *O. mykiss* density estimates from 2007-2015. Data from 2006 are not comparable due to differences in sampling methods, and thus are not included.

	O. mykiss/100m² and 95% confidence interval								
Reach	2007	2008	2009	2010	2011	2012	2013	2014	2015
Alder Springs	48 ± 28	24 ± 24	12 ± 4	11 ± 4	24 ± 5	7 ± 2	27 ± 2	25 ± 4	36 ± 4
Road 6360	25 ± 10	9 ± 3	24 ± 9	13 ± 4	15 ± 3	15 ± 3	31 ± 3	15 ± 6	15 ± 3
Camp Polk	60 ± 13	52 ± 21	57 ± 15	27 ± 9	16 ± 3	-	-	-	-
Wolftree-Aspen Hall	-	-	21 ±7	106 ± 29	42 ± 9	32 ± 9	-	31 ± 7	10 ± 2
Sisters	20 ± 10	5 ± 2	23 ± 14	18 ± 6	10 ± 5	28 ± 8	27 ± 8	35 ± 6	59 ± 7
USFS site at TSID	-	-	2.4 (1.5-4.0)[2]	-	-	1.7 (0.8-4.2)[3]	-		

[2] 95% Confidence Interval as reported in Reischauer and Dachtler 2009.
[3] Personal communication, M. Riehle, November 2016

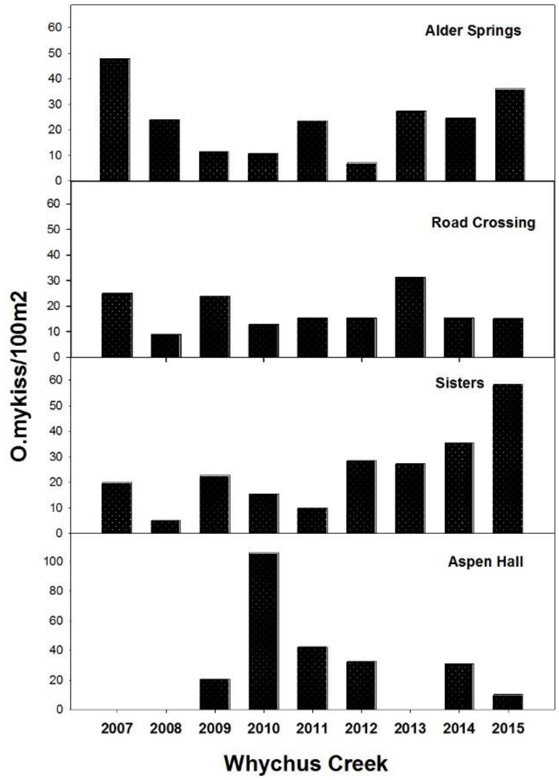

Figure 4. *O. mykiss* densities at four Whychus Creek sites from 2007 to 2015. Surveys at Camp Polk were discontinued following diversion of Whychus Creek into the restored channel. Reproduced with permission from Madden *et al* 2016.

Spring Chinook juvenile density & size

Consistent with no Chinook fry having been released into Whychus Creek in 2015, no juvenile Chinook were captured during electrofishing surveys in Whychus Creek. Juvenile spring Chinook densities in Whychus Creek were low in 2013, but the total number caught was higher than in previous years (Table 4, Figure 5). Chinook captured in Whychus Creek in 2013 were significantly larger than those captured in 2009 and 2010, but were similar in size to Chinook captured in 2011 and 2012.

Table 4. Spring Chinook densities in Whychus Creek in 2009-2013 estimated from mark-recapture and snorkel surveys. No Chinook were released, and accordingly none captured, in Whychus Creek in 2014 or in 2015.

	Spring Chinook Density (Fish/100m^{2})							
	2009		2010		2011		2012	2013
Reach	Mark Recapture	Snorkel	Mark Recapture	Snorkel	Mark Recapture	Snorkel	Mark Recapture	
Alder Springs	6 ± 5.8	--	0	--	4 ± 2	--	1.8 ± 0.9	3 ± 1
Road 6360	4.7 ± 2.3	8	1 ± 0.5	2	2 ± 1	0	1.2 ± 0.5	1 ± 0.5
Camp Polk	17.4 ± 14.7	--	0	--	0	--	--	--
Wolftree-Aspen Hall	8.3 ± 6.1	0	25 ± 16	4	15 ± 19	1	0.8 ± 0.0	--
Sisters	2 ± 0.8	4	0	4	0	--	10.1 ± 7.9	4 ± 1

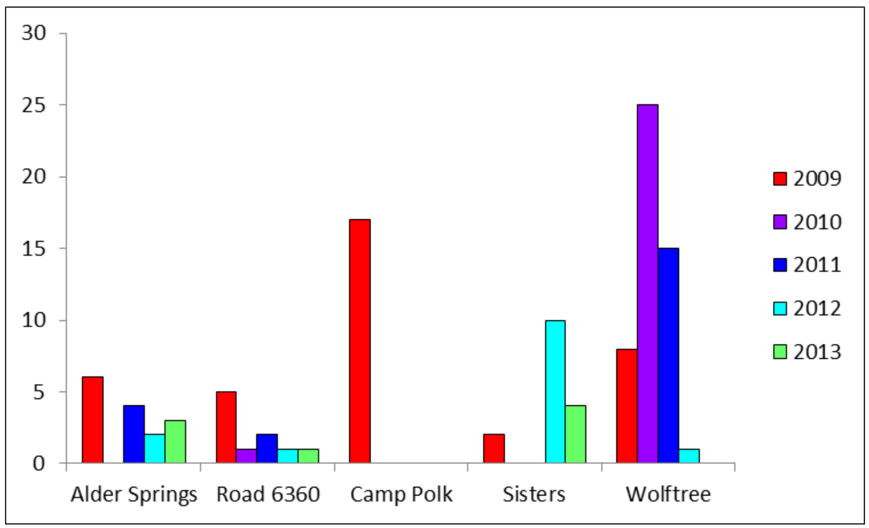

Figure 5. Juvenile Spring Chinook densities in Whychus Creek in 2009-2013 estimated from mark-recapture surveys

Juvenile migration and smolt production

Three hundred twenty-nine (329) naturally-reared steelhead smolts were trapped and released on Whychus Creek in 2015. Low trap efficiency for naturally reared steelhead, estimated at 5.2%, yielded a population estimate of 6,223 ± 1,797 naturally reared steelhead migrating out of Whychus. Naturally-reared steelhead smolts were captured on Whychus between March and June, with migration peaking in May.

Hatchery-reared steelhead released in Whychus on April 13 were captured at the screw trap between April 14 and May 25. The same 5.2% trap efficiency applied to the 1,815 hatchery-reared steelhead captured at the trap results in a hatchery-reared smolt migration estimate of 34,904, higher than the actual number released, suggesting the majority of hatchery-reared steelhead released as smolts migrated out of Whychus Creek. This number also indicates trap efficiency was underestimated. 2015 is the first year for which a sufficient number of smolts were trapped on Whychus Creek or the Deschutes Arm of Lake Billy Chinook to generate a smolt production estimate.

Hatchery-reared Chinook released in Whychus on March 9 migrated out of the creek quickly, with over 99% captured at the screw trap within a week of release. No Chinook fry were released in Whychus in 2015 and accordingly no naturally-reared Chinook salmon were captured at the Whychus Creek screw trap in 2015.

Adult returns and migration

Ninety-three adult steelhead were captured at the Pelton Trap between September 2014 and April 2015, with captures peaking in October and early January (Burchell et al 2016). Of these, 73 originated from fry releases or natural production (right maxillary-clipped) and 20 from smolt releases (left maxillary-clipped). Fifty-two adult Chinook salmon, including 38 fish originating from fry releases and 14 from smolt releases, returned to the Pelton Trap between early May and August 2015, peaking in late May. The number of adult steelhead and Chinook salmon returning to the Pelton Trap each year has fluctuated since the first fish returned in 2012 (Table 4). Ninety of the adult steelhead were JCART-tagged, and all 93 were double-tagged and released upstream of Round Butte Dam; 51 adult chinook were double- and JCART-tagged and released upstream of the dam.

Four steelhead (4%) ascended the Deschutes River Arm from Lake Billy Chinook and subsequently ascended Whychus Creek. One (260-22) was mobile tracked to the mouth of Whychus Creek but not otherwise detected; one (920-39) swam back down the Deschutes River and was last detected at the confluence of the Deschutes River and Lake Billy Chinook; one (920-54) swam back into the Deschutes River and was last detected at Scout Camp; and one (920-201) swam back into Lake Billy Chinook, up the Crooked River, and was last detected at Opal Springs.

Seven Chinook (14%) ascended the Deschutes River Arm of Lake Billy Chinook, and six continued up to the confluence with Whychus Creek. Four of the seven ascended the Crooked River before ascending the Deschutes River Arm; one swam from the confluence of the Deschutes and Whychus to the Crooked River and back to the Deschutes. One of the seven was detected in Whychus Creek, and one swam past Whychus Creek and was detected in the Deschutes River approximately 3.7 rkm above Whychus.

No returning steelhead or Chinook ascended Whychus Creek in 2014. In 2013 one steelhead and three Chinook ascended Whychus Creek as high as Rimrock Ranch.

Table 4. Returning adult steelhead and Chinook salmon captured at the Pelton Trap 2012-2015

	Steelhead			Chinook		
	Adults captured at Pelton Trap	Originating from fry releases	Originating from smolt releases	Adults captured at Pelton Trap	Originating from fry releases	Originating from smolt releases
2012	--	--	--	50	no data[4]	no data
2013	133	104 (78%)	29 (22%)	22	12 (55%)	10 (45%)
2014	50	38 (76%)	12 (24%)	24	20 (83%)	4 (17%)
2015	93	73 (78%)	20 (22%)	52	38 (73%)	14 (27%)

O. mykiss redd surveys

Surveyors identified a total of 34 redds in Whychus Creek in 2015 (Madden and Bennett 2016; Table 5). As in previous years, a large majority of redds (88%) were located in reaches 1-3, in the vicinity of Alder Springs. Redds averaged 6.5/km in mainstem sites, down from 9.5/km in 2014. Spawning in Whychus was recorded from March through July and peaked in May.

[4] Data not reported in PGE 2012 Adult Migration report (Hill and Quesada 2013)

Table 5. *O. mykiss* redd survey sites, reaches, and count data, 2006-2015. Surveys at RKM 26 were discontinued in 2012; because only one Camp Polk site was surveyed 2012-2015, no total is reported as it would not be comparable to totals reported for two Camp Polk sites from 2006-2011. (NC = Not Comparable).

Site Description	Site Type	Number of Redds									
		2006	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**
RKM 2	Index	14	51	27	4	16	21	7	14	30	28
RKM 27	Index	14	4	3	3	3	3	1	2	4	1
Lewis Woodpecker Creek	Trend (Original)	6	1	1	6	2	0	1	4	2	1
Alder Springs Creek	Trend (Original)	3	12	9	1	14	0	0	2	2	1
RKM 25	Trend (Original)	8	6	0	4	3	4	6	2	4	2
Upstream of Alder Springs Creek	Original	4	5	11	13	4	--	0	--	--	--
Rimrock Ranch/RKM 14	Random (2012) (Original)	35	38	18	10	--	--	0	--	--	--
RKM 26	Original	9	11	5	2	8	4	--	--	--	--
Upstream of Road 4606 Footbridge	Original	9	0	0	0	--	--	--	--	--	--
RKM 38	Random (2012) (Original)	10	1	0	0	--	--	0	--	--	--
RKM 8	Random (2010)	--	--	--	--	3	--	--	--	--	--
RKM 11	Random (2011)	--	--	--	--	--	3	--	--	--	--
RKM 21	Random (2013, 2014)	--	--	--	--	--	--	--	1	0	--
RKM 22	Random (2015)	--	--	--	--	--	--	--	--	--	0
RKM 23	Random (2010)	--	--	--	--	12	--	--	--	--	--
RKM 34	Random (2011)	--	--	--	--	--	6	--	--	--	--
RKM 36	Random (2015)	--	--	--	--	--	--	--	--	--	1
Alder Springs (RKM 2, Lewis Woodpecker and Alder Springs Creeks)		**23**	**64**	**37**	**11**	**32**	**21**	**8**	**20**	**34**	**30**
Camp Polk (RKM 26 and 27)		**23**	**15**	**8**	**5**	**11**	**7**	NC	NC	NC	NC
TOTAL		**112**	**129**	**74**	**43**	**65**	**41**	**15**	**25**	**42**	**34**

Discussion

O. mykiss and Chinook salmon population estimates

Estimated juvenile *O. mykiss* density bumped up at two Whychus Creek sites in 2015 (Madden *et al* 2016). Juvenile density in 2015 was almost twice as high at Sisters as in any other year, and density at Alder Springs was higher than in any other year except 2007. Density at Road Crossing was similar to most other years, and at Wolftree-Aspen Hall was lower than in any year. On average, the highest juvenile densities have been observed at Alder Springs, followed by Sisters, the 6360 Road Crossing, and Wolftree. The consistently relatively higher densities at Sisters suggest this reach may provide suitable rearing habitat; particularly so in combination with cooler summer stream temperatures in this reach than in downstream reaches. The marked increase in O. mykiss at Sisters in 2015 coincides with the first year of sampling following the Whychus Floodplain stream channel and floodplain restoration project, which created extensive side- and off-channel habitat. Despite continued low *O. mykiss* densities in Whychus, juvenile density estimates remain approximately 7x the 4-5 fish/100m^2 average of redband densities reported for studies completed prior to steelhead reintroduction in 2007 (Dachtler 2007, Riehle and Lovtang 2000, Groves *et al* 1999), consistent with the continued annual releases of steelhead fry. Although no juvenile density estimate was available for Chinook in Whychus Creek in 2015 or 2014, Chinook densities in previous years were lower in Whychus Creek than in other rivers and tributaries sampled (Metolius, Lake Creek, Ochoco Creek).

Several hypotheses exist for the continued low density of juvenile *O. mykiss* and spring Chinook salmon, and low adult returns, in Whychus Creek. Degraded stream habitat, specifically low availability of off-channel habitat and habitat complexity to provide refuge, may result in high juvenile mortality and/or fry flushing out of the creek during high flow events. Chinook smolt outplants were observed to move out of Whychus Creek within 24 hours of release in 2013 and 2015, suggesting they may not stay in the creek long enough to imprint on Whychus as their home stream and therefore would not return to the tributary as adults. Chinook and steelhead released above the Pelton-Round Butte facility into Lake Billy Chinook may not be able to detect the chemical signature of Whychus or the Deschutes in the mixed currents of the reservoir; alternatively, Lake Billy Chinook currents may convey returning adults from the release site toward the Crooked River, requiring returning adults to detect and actively swim toward the Deschutes.

Genetic analysis of 2005 (pre-reintroduction) and 2013 (post-reintroduction) Whychus Creek juvenile *O. mykiss* populations showed *O. mykiss* in Whychus in 2013 to be predominately (75%) descended from Round Butte Hatchery stock, with natural-origin *O. mykiss* largely absent (15%) from the 2013 Whychus Creek population (Adams *et al* 2015). Although this finding suggests Round Butte Hatchery O. mykiss far outnumber natural-origin O. mykiss in Whychus, these percentages are not incompatible with the increase in density estimates since reintroduction, i.e. the low percentage of natural-origin Whychus Creek O. mykiss may be consistent with the low numbers of redband found in surveys prior to reintroduction. The high percentage of Round Butte Hatchery O. mykiss apparently residualizing in the creek may have future implications for the contribution of a Whychus Creek population to Mid-Columbia steelhead recovery and for expression of genetic and life-history diversity in the Whychus Creek *O. mykiss* population.

Studies on life history plasticity in salmonids suggest the life history pathway exhibited by an individual fish is determined by the environmental context, and that the genetic threshold that cues one pathway or another varies as a function of local adaptation (Sogard et al 2012). In the Mokelumne River in

California's Central Valley, *O. mykiss* growth rate as a function of time of emergence and potentially stream temperature and fish density resulted in a high proportion of *O. mykiss* adopting a resident life history, relative to *O. mykiss* in the American River that experienced warmer stream temperatures and potentially lower densities, and exhibited higher growth rates and uniform adoption of an anadromous life history (Sogard et al 2012). These results suggest that growth rate as a function of density, stream temperature, and food availability may be contributing to presumed high rates of *O. mykiss* residualization in Whychus Creek.

Life histories and restoration needs in Whychus

Redd counts, juvenile outmigration trapping, and radio tracking adult steelhead and Chinook salmon following their release upstream of the Round Butte dam are providing data that allow preliminary description of life history timing and associated locations of life history activities for resident redband and the two reintroduced species (Appendix A). This knowledge in turn suggests when and where it is most critical for restoration partners to focus efforts to restore sufficient conditions that will support each life history. Redband and steelhead rearing occur year-round in Whychus, requiring 18°C stream temperatures year-round. Redd counts conducted between 2007 and 2015 show the greatest concentration of redds consistently located in the Alder Springs reaches and at Rimrock Ranch. Steelhead spawning is anticipated to occur in April and May in Whychus with Chinook spawning from August to October, recommending flow management that will maintain temperatures below 13°C at Alder Springs and Rimrock Ranch during these months. New stream habitat data from completed stream channel and floodplain restoration projects may provide insight into the value of habitat created through these projects for relevant life histories of summer steelhead and spring Chinook salmon.

Conclusions

O. mykiss and spring Chinook study data from Whychus Creek and the Deschutes River Arm of Lake Billy Chinook cumulatively depict low juvenile abundance, low rates of spawning, and few adult returns. Survey data from 2015 are consistent with previous years' findings showing low *O. mykiss* densities and no statistically significant trend in *O. mykiss* abundance. 2015 redd count numbers remained higher than in 2013 and 2012 but were lower than in all other years since 2006. Four returning steelhead and seven returning Chinook ascended the Deschutes from Lake Billy Chinook in 2015, one of each of which entered Whychus.

Genetic analyses show that seven years (2007-2013) of releasing Round Butte Hatchery *O. mykiss* into Whychus Creek have dramatically increased the proportion of the Whychus O. mykiss population comprised of Round Butte Hatchery stock, suggesting some percentage of released fry are residualizing in the creek. A low percentage of natural-origin fish may reflect low initial juvenile O. mykiss densities prior to reintroduction. PGE will conduct further analyses to differentiate between juvenile redband and steelhead five and ten years (in 2017 and 2022, respectively) after returning steelhead were first passed upstream of the dams in 2012. Additional genetic analyses will allow researchers to support preliminary findings and better understand the status and trends of natural-origin redband and reintroduced steelhead populations in Whychus Creek and other Upper Deschutes Basin rivers and tributaries.

Information on seasonal food availability and stream temperature in Whychus Creek and Crooked River tributaries could be used with fisheries monitoring data, specifically growth rates, to better understand patterns of life history pathways in reintroduced steelhead in the Upper Deschutes basin. Ongoing refinements of PGE native fish monitoring techniques will continue to improve information available, specifically numbers of smolts outmigrating from Whychus, and contribute to a better understanding of the fate of outplanted steelhead fry as well as factors, including stream conditions, influencing *O. mykiss* density, production, and life history adoption in Whychus Creek.

Restoration partners initially expected that biological indicators would provide an effective means for evaluating trends in watershed restoration. In the short term, data available on *O. mykiss* and Chinook salmon in Whychus Creek are insufficient to evaluate how restoration may be influencing population trends for either reintroduced anadromous species or for native resident fish. Over the long term, as stream conditions stabilize following restoration, adult steelhead and Chinook salmon return to spawn in Whychus, and steelhead and Chinook releases are ultimately replaced by natural spawning runs, fish population trends will more directly reflect stream habitat and watershed conditions. When these criteria are met, fish population data may provide a more useful indicator of restoration effectiveness.

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APPENDIX A. Preliminary life history timing and temperature requirements of redband, steelhead, and Chinook salmon in Whychus Creek.

		Temp	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Redband	Spawning[a]	18°C												
	Egg Incubation & Emergence													
	Juvenile Rearing													
Steelhead	Spawning[a]	12.8°C												
	Egg Incubation & Emergence													
	Juvenile Rearing[b]	18°C												
	Juvenile Outmigration[c]													
	Adult In-migration[d]													
Chinook	Spawning[e]	12.8°C												
	Egg Incubation & Emergence[e]													
	Juvenile Rearing[e]	18°C												
	Juvenile Outmigration[c, e]													
	Adult In-migration[d, e]													

- Peak activity
- Activity documented/known
- Predicted activity

[a] Source: PGE redd count data, 2007-2013
[b] Quinn 2005
[c] Hill and Quesada 2014
[d] Hill et al. 2014
[e] B. Spateholts 2015