Upper Deschutes Watershed Council
Technical Report

2013 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
EPA	Environmental Protection Agency
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
CI	Confidence Interval
df	Degrees of freedom
DO	Dissolved oxygen
°F	Fahrenheit
mg/L	Milligrams per liter
OAR	Oregon Administrative Rules
QA/QC	Quality assurance / quality control
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 to 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 saw the 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 selected 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 completed 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 the UDWC focuses on tracking the status and trends of key physical and biological indicators in Whychus Creek. The UDWC selected these indicators based on a conceptual model of factors limiting salmonid production in the creek (Figure 1). They expect 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. The UDWC 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 2013 Whychus Creek Monitoring Report assesses indicators in one of these categories.

From 2011 to 2012 UDWC collaborated with Bonneville Environmental Foundation and Deschutes Partnership organizations to refine the Whychus Creek Model Watershed conceptual model using The Nature Conservancy’s Conservation Action Planning framework (TNC 2007). This process identifies conservation values to be ameliorated through restoration actions, direct threats to conservation values resulting from human activities, root causes driving those threats, strategic actions to address threats, and linkages among these elements to define and examine assumptions about both the specific mechanisms contributing to habitat degradation and the actions identified to restore conservation values. This model and the specific strategies and actions that were developed through the planning process, summarized in the Whychus Creek Watershed Restoration Plan Update (UDWC 2013), update and elaborate on the 2009 Whychus Creek Watershed Monitoring Plan (UDWC 2009).

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.

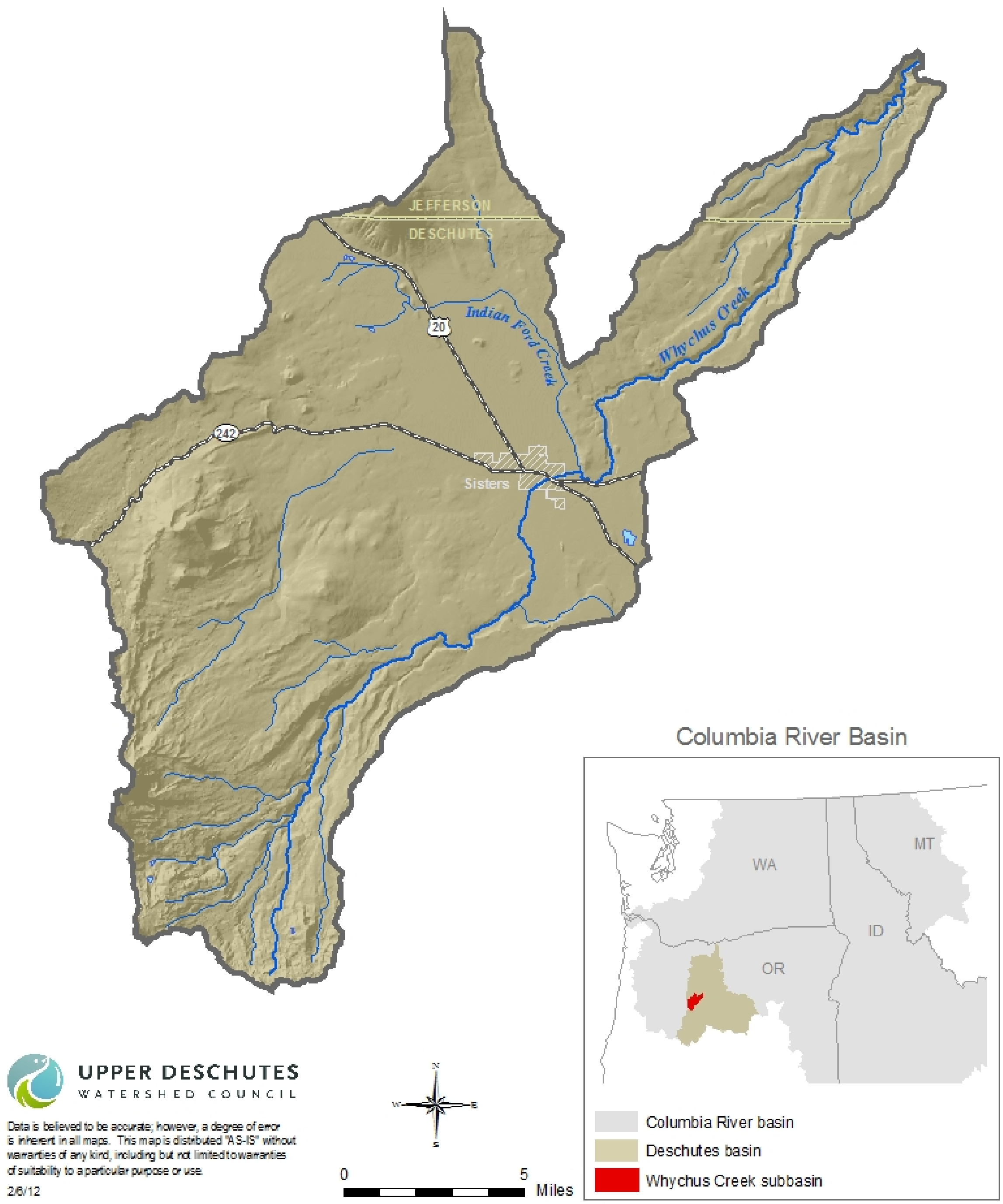

Figure 2.
Whychus Creek extends from the Cascade Range to the Deschutes River. The creek's watershed encompasses approximately 162,000 acres of Deschutes and Jefferson Counties in central Oregon.

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 realignment, fish passage improvements, screening of irrigation diversions, and other restoration actions. Baseline conditions are reported in the 2009 Whychus Creek Monitoring Report (UDWC 2010). These conditions were inventoried following some streamflow restoration but prior to any other restoration efforts.

Golden & Wymore (2014) document summer stream flow conditions in Whychus Creek from 2000-2013. This report focuses on metrics representing low flow conditions in the creek. Mork (2014a) answers questions related to stream temperature in Whychus Creek, drawing on thirteen years of data to examine water quality in relation to state standards and to stream flow restoration. Restored stream flow has affected metrics in each of these reports.

Two reports quantify habitat improvements anticipated to result from restoration projects completed subsequent to baseline analyses. Mork (2014b) documents the status of fish passage barriers as a measure of stream connectivity along the creek. Restoration partners expect to provide passage at each of the original six barriers identified in the 2009 report. Mork (2013c) discusses 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, with the expectation that restoration partners will screen each of these diversions in the future. No new habitat surveys were conducted on Whychus Creek in 2013; accordingly this report does not include an updated habitat quality study.

Two additional reports update the status of biological conditions in the creek. Mork (2014d) summarizes PGE's 2013 steelhead and chinook survey results. This report outlines the status of juvenile fish, spawning redds, and migration in the creek, and discusses how future sampling and analysis will expand the current understanding of these populations. Although the ongoing reintroduction of steelhead and chinook in Whychus Creek prevents meaningful application of population data for these species as an indicator of restoration effectiveness, tracking fish population trends in Whychus provides essential information for future evaluation of reintroduction and restoration efforts. Mazzacano (2014) examines five years of macroinvertebrate data to identify trends in macroinvertebrate community composition before and after extensive streamflow restoration.

These six reports evaluate improvements in stream conditions in 2013 as measured by the status of physical and biological indicators subsequent to major streamflow and channel restoration and irrigation diversion retrofits. The reports and the data they contain will help restoration partners to understand the effectiveness of their actions at moving the creek toward desired conditions. Restoration partners continue to draw from these reports to continually improve restoration implementation and monitoring in Whychus Creek.

References

Bash JB, Ryan CM. 2002. Stream restoration and enhancement projects: is anyone monitoring? Environmental Management. 29: 877-885.

Federal Energy Regulatory Commission (FERC). 2005. Order Approving Settlement and Issuing New License. Project No. 2030-036. Document Number 20050621-3052.

Gannett MW, Lite Jr KE, Morgan DS, Collins CA. 2001. Groundwater Hydrology of the Upper Deschutes Basin, Oregon. Water Resources Investigations Report 00-4126. Portland, OR: United States Geological Survey.

Golden B, Wymore K. 2014. "Whychus Creek Stream Flow." Pages 8-19 in Mork L, Houston R, Editors. 2013 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mazzacano C. 2014. Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, and 2011-2013. Pages 61-102 in Mork L, Houston R, Editors. 2013 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mork L. 2014a. "Whychus Creek Water Quality Status, Temperature Trends, and Stream flow Restoration Targets." Pages 20-46 in Mork L, Houston R, Editors. 2013 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mork L. 2014b. "Stream Connectivity in Whychus Creek." Pages 47-53 in Mork L, Houston R, Editors. 2012 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mork L. 2014c. "Fish Entrainment Potential in Whychus Creek." Pages 54-60 in Mork L, Houston R, Editors. 2013 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mork L. 2014d. "Native Fish Monitoring in Whychus Creek." Pages 103-121 in Mork L, Houston R, Editors. 2013 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Nehlsen W. 1995. Historic Salmon and Steelhead Runs of the Upper Deschutes River and Their Environments. Portland, Oregon: Report to Portland General Electric.

O'Donnell TK, Galat DL. 2008. Evaluating success criteria and project monitoring in river enhancement within an adaptive management framework. Environmental Management. 42: 90-105.

Souchon Y, Sabaton C, Deibel R, Reiser D, Kershner J, Gard M, Katapodis C, Leonard P, Poff NL, Miller WJ, Lamb BL. 2008. Detecting biological responses to flow management: missed opportunities; future directions. River Research and Applications. 24: 506-518.

TNC (The Nature Conservancy). 2007. Conservation Action Planning: Developing strategies, taking action, and measuring success at any scale; Overview of Basic Practices. 18 p. www.conservationgateway.org/cap

UDWC (Upper Deschutes Watershed Council). 2009. Whychus Creek Restoration Monitoring Plan. Upper Deschutes Watershed Council. Bend, OR.

USFS (United States Forest Service). 1998. Sisters / Whychus Watershed Analysis. Sisters Ranger District, USFS. Sisters, OR.

Whychus Creek Stream Flow

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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. They occurred during late May in 2011 and 2012. In 2013, they occurred during late July. Annual minimum 30 day moving average flows increased or remained constant in every year except for 2005 and 2009. 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 high of 32 cfs in 2011. 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. These results highlight the need to understand whether low flows during these periods limit ecosystem function and, if so, to focus on restoration efforts during these periods. 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. Human disturbances also prevent natural disturbances from occurring. Both of these 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.

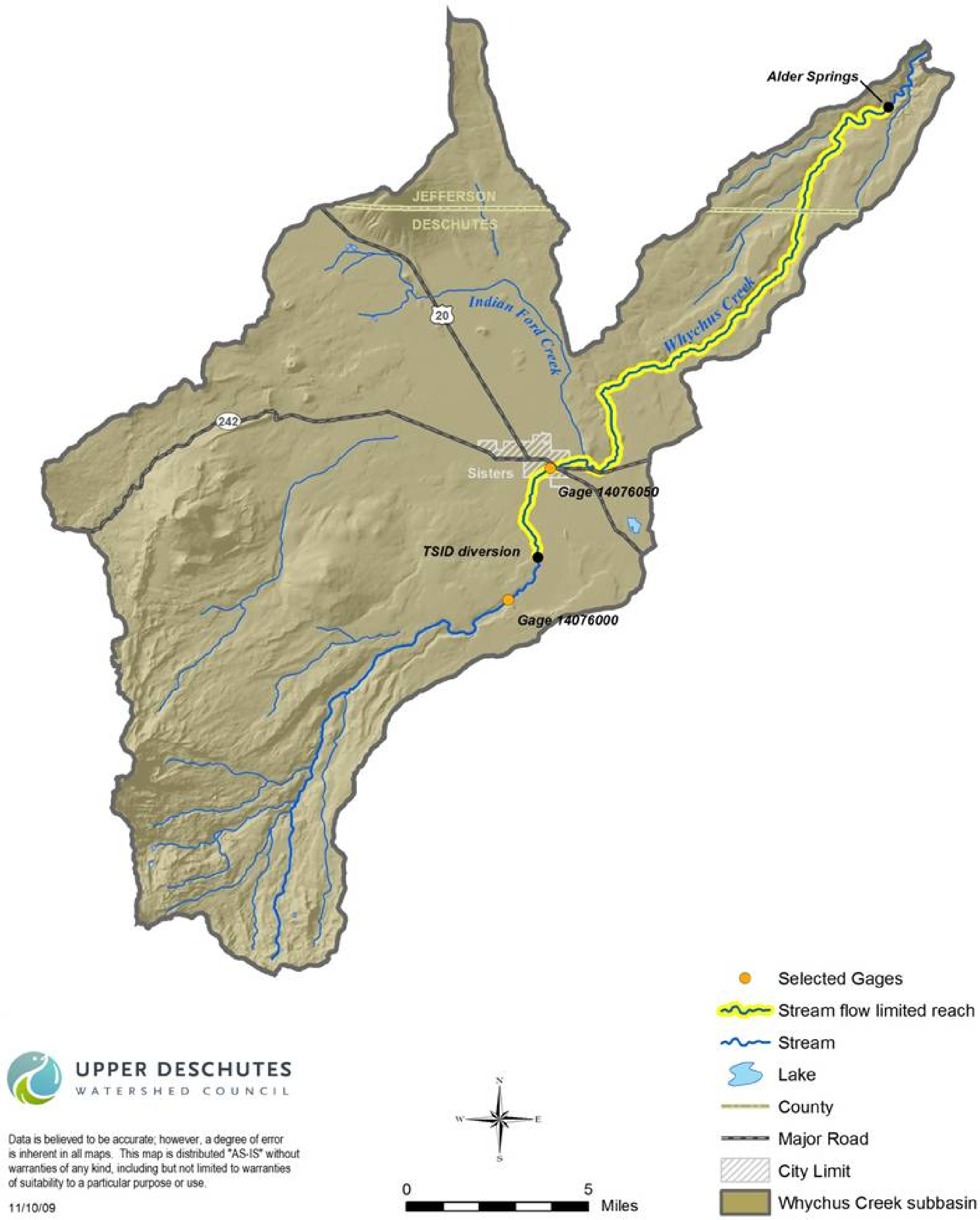

Figure 1.
Stream flow limits stream function in Whychus Creek downstream from the Three Sisters Irrigation District Diversion. Spring inputs near the mouth of Whychus Creek increase stream flow and improve conditions in the creek.

Metric	Appears In
30 day minimum	Gao et al2009, Richter et al1996
May median flow	Gao et al2009, Richter et al1996
August median flow	Richter et al1996

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 2013.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 2013. 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 2013 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-realtime 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 2014). OWRD reviewed this data again before publishing it as daily average discharge data online (OWRD 2013). 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 and from October 1, 2011 through October 31, 2013 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 2013. 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.

Minimum 30 Day

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 2013.

May Median

The DRC used spreadsheet software to determine the median daily average flow during the month of May for years 2001 through 2013. The DRC only had partial data for 2000 and did not include that data in this analysis.

<u>August Median</u>

The DRC used spreadsheet software to determine the median daily average flow during the month of August for years 2000 through 2013. The DRC had full 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 19.00 cfs in 2010. It increased or remained constant each year except for 2005 and 2009.

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

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.

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 2). 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.

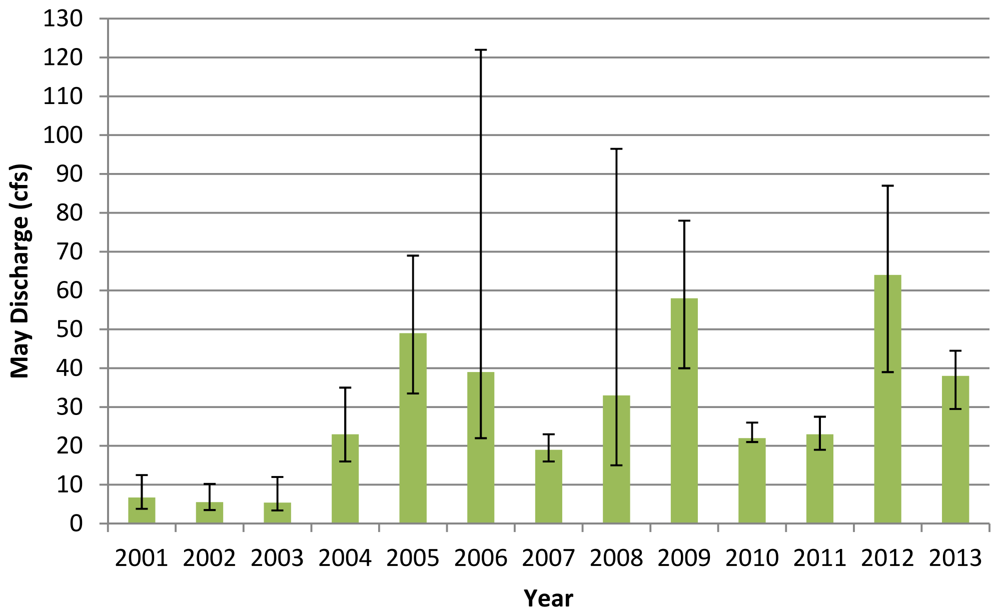

Figure 2.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of May provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of May at this location.

August Median

Median discharge during the month of August exhibited both inter-annual and intra-annual variation at gage number 14076050 (Figure 3). 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.

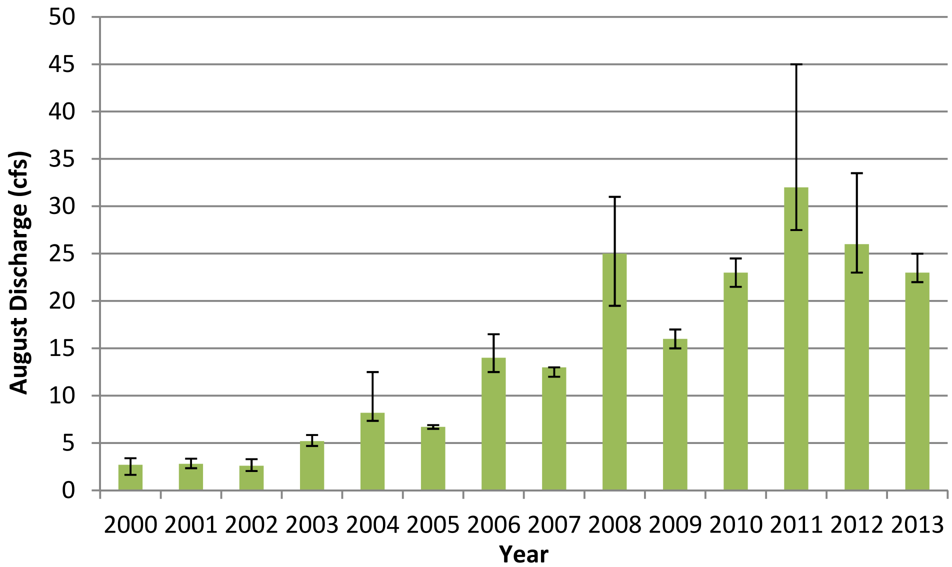

Figure 3.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of August provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of August at this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

Discussion

The analyses in this report describe baseline stream flow conditions in Whychus Creek over a fourteen-year period of intensive restoration. They focus on the period from 2000 through 2013. 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 than in the past. Minimum 30-day moving average data suggest that flow lows now occur more often in late spring/early summer.

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.

The Oregon Department of Fish and Wildlife applied for and received instream water rights in the 1990s to support fish populations. These water rights provide one set of base flow targets. 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 seven out of thirteen years (OWRD 1996, Figure 2). It never met Oregon's March, April, and May instream water right of 50 cfs for Whychus Creek downstream from Indian Ford Creek (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 may also be important for stream function. As noted earlier, redband trout spawning centers on the month of May (ODFW 2005). Consistently low stream flow and corresponding high temperatures during late April, May, and early June may limit available spawning habitat. Extreme low flow events during this period may limit fish production by dewatering existing redds. Results suggests that extreme low flows are occurring less often now 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 each year included in this study through 2010 (Table 2). Stream flow naturally decreases during this period, so periodically low late summer and early fall flows do not necessarily limit stream functions. However, average August stream flow under natural conditions has been calculated to be 100 cfs (OWRD 2011).

The magnitude and frequency of current late summer and early fall flows in Whychus Creek, suggest that low flows may limit fish populations Temperature-flow analyses of July temperature data from Whychus Creek show that stream flows of at least 60 cfs are required to meet Oregon's temperature standards for salmon and trout rearing and migration during July at the warmest locations on the creek (Mork 2014). These analyses support the conclusion that flows of less than 60 cfs may yield temperatures that limit fish populations.

Instream water rights applied for and received by the Oregon Department of Fish and Wildlife again provide a rough base flow target in Whychus Creek. Median daily average flows during the month of August exceeded Oregon's 20 cfs instream water right for Whychus Creek upstream from Indian Ford Creek in 2008, 2010, 2011, 2012 and 2013. (OWRD 1996a, Figure 3). They 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. Increasing these flows should remain a priority for restoration partners and they should continue to use August or September median flows as an indicator of restoration effectiveness.

Conclusions

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 applied for by the Oregon Department of Fish and Wildlife in the 1990s as stream flow targets. Based on the findings of Mork (2014), target stream flows will not yield sufficiently low stream temperatures to meet Oregon's temperature standards for trout and salmon spawning, rearing and migration at some locations and some times of year.

Legally protected instream water rights currently meet or exceed these targets in some locations and times of year. The reliability of these water rights instream varies based on water availability in Whychus Creek and Oregon's prior appropriation system of allocating water, though, and actual stream flows do not always meet the legally protected instream water rights. These conditions contribute 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 restoration actions correlate with stream flow outcomes in Whychus Creek, particularly during early summer. Evaluating additional extreme low flow metrics may further inform restoration partners as to the success of their actions.

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.

References

Bash JB, Ryan CM. 2002. Stream restoration and enhancement projects: is anyone monitoring? Environmental Management. 29: 877-885.

Decker AS, Bradford MJ, Higgins PS. 2008. Rate of biotic colonization following flow restoration below a diversion dam in the Bridge River, British Colombia. 24: 876-883.

Dewson ZS, James ABW, Death RG. 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. Journal of the North American Benthological Society. 26: 401-415.

Doyle MW, Stanley EH, Strayer DL, Jacobson RB, Schmidt JC. 2005. Effective discharge analysis of ecological processes in streams. Water Resources Research. 41: W11411.

Federal Energy Regulatory Commission (FERC). 2005. Order Approving Settlement and Issuing New License. Project No. 2030-036. Document Number 20050621-3052.

Gannett MW, Lite JrKE, Morgan DS, Collins CA. 2001. Groundwater Hydrology of the Upper Deschutes Basin, Oregon. Water Resources Investigations Report 00-4126. Portland, OR: United States Geological Survey.

Gao Y, Vogel RM, Kroll CN, Poff NL, Olden JD. 2009. Development of representative indicators of hydrologic alteration. Journal of Hydrology. 374: 136-147.

Helsel DR and Hirsch RM. 2002. Statistical Methods in Water Resources. In Techniques of Water-Resources Investigations of the United States Geological Survey.Book 4, Hydrologic Analysis and Interpretation. Chapter A3. United States Geological Survey.

James ABW, Dewson ZS, Death RG. 2008. The effect of experimental flow reductions on macroinvertebrate drift in natural and streamside channels. River Research and Applications. 24: 22-35.

Konrad CP, Brasher AMD, May JT. 2008. Assessing stream flow characteristics as limiting factors on benthic invertebrate assemblages in streams across the western United States. Freshwater Biology. 53:1983-1998.

MacDonald LH, Smart AW, Wissmar RC. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. U.S. Environmental Protection Agency Region 10, NPS Section. Seattle, WA.

Milly PCD, Betancourt J, Falkenmark M, Hirsch RM, Kundzewicz ZW, Stouffer RJ. 2009. Stationarity is dead: wither water management? Science. 319: 573-574.

Monk WA, Wood PJ, Hannah DM, Wilson DA, Extence CA, Chadd RP. 2006. Flow Variability and macroinvertebrate community response within riverine systems. River Research and Applications. 22: 595-615.

Mork LA. 2014. Whychus Creek Water Quality Status, Temperature Trends, and Stream Flow Restoration Targets. Upper Deschutes Watershed Council, Bend, Oregon. 26 p.

Nehlsen W. 1995. Historic Salmon and Steelhead Runs of the Upper Deschutes River and Their Environments. Portland, Oregon: Report to Portland General Electric.

NRC (National Research Council). 2002. Upstream: Salmon and Society in the Pacific Northwest. Committee on Protection and Management of Pacific Northwest Anadromous Salmonids, National Research Council, National Academies. Washington, DC.

O'Donnell TK, Galat DL. 2008. Evaluating success criteria and project monitoring in river enhancement within an adaptive management framework. Environmental Management. 42: 90-105.

ODFW (Oregon Department of Fish and Wildlife). 2005. Unpublished life history periodicity chart for salmonids in the lower Deschutes River. Bend, Oregon: Oregon Department of Fish and Wildlife.

Olden JD, Poff NL. 2003. Redundancy and the choice of hydrologic indices for characterizing stream flow regimes. River Research and Applications. 19:101-121.

OWRD (Oregon Water Resources Department). 1996a. Certificate of Water Right Number 73224. Salem, OR: Oregon Water Resources Department.

OWRD (Oregon Water Resources Department). 1996b. Certificate of Water Right Number 73223. Salem, OR: Oregon Water Resources Department.

OWRD (Oregon Water Resources Department). 2011. Whychus Creek Flow Diagram: August Typical Flows. Salem, OR: Oregon Water Resources Department.

OWRD (Oregon Water Resources Department). 2013. Data for Gage 14076050, Whychus Creek at Sisters, OR. May 18, 2000 through October 31, 2013.

OWRD (Oregon Water Resources Department). 2014. Surface Water Data Development Process. Salem, OR: Oregon Water Resources Department. Available at http://www.wrd.state.or.us/OWRD/SW/about_data.aspx Accessed July 17, 2014.

Pyrce R. 2004. Hydrological low flow indices and their uses. WSC Report 04-2004. Ontario, Canada: Watershed Science Centre.

Richter DB, Baumgartner JV, Powell J, Braun DP. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology. 10: 1163-1174.

Roni P. 2005. Monitoring Stream and Watershed Restoration. American Fisheries Society, Bethesda, Maryland. 350 pp.

Souchon Y, Sabaton C, Deibel R., Reiser D., Kershner J, Gard M, Katapodis, C, Leonard P, Poff NL, Miller WJ, Lamb BL. 2008. Detecting biological responses to flow management: missed opportunities; future directions. River Research and Applications. 24: 506-518.

Stromberg JC, Bagstad KJ, Leenhouts JM, Lite SJ, Makings E. 2005. Effects of stream flow intermittency on riparian vegetation of a semiarid region river (San Pedro River, Arizona). River Research and Applications. 21: 925-938.

Thompson K. 1972. Determining stream flows for fish life. Presentation at Pacific Northwest River Basins Commission Instream Flow Workshop. March 15-16, 1972.

UDWC (Upper Deschutes Watershed Council). 2002.Whychus Creek Watershed Action Plan. Upper Deschutes Watershed Council. Bend, OR.

UDWC (Upper Deschutes Watershed Council). 2008. Whychus Creek Restoration Monitoring Plan. Upper Deschutes Watershed Council. Bend, OR.

USFS (United States Forest Service). 1998. Sisters / Whychus Watershed Analysis. Sisters Ranger District, U.S. Forest Service. Sisters, OR.

Wills TC, Baker EA, Nuhfer AJ, and Zorn TG. 2006. Response of the benthic macroinvertebrate community in a northern Michigan stream to reduced summer stream flows. River Research and Applications. 22: 819-836.

Yang YE, Cai X, Herricks EE. 2008. Identification of hydrologic indicators related to fish diversity and abundance: a data mining approach for fish community analysis. Water Resources Research. 44. W04412

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

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Abstract

Diversion of almost 90% of summer streamflow and channelization of over 50% of the length of Whychus Creek have degraded water quality, leading to an ODEQ listing of water quality limited in 2002, 2004, and 2010. The Upper Deschutes Watershed Council monitored temperature from 1995 through 2013 at eleven sites representing diverse flow conditions in Whychus Creek. This report incorporates 2013 data to 1) evaluate the current status of temperature in Whychus Creek in relation to state standards for salmonid spawning, rearing and migration; and 2) refine target flows projected to produce temperatures that meet state standards. Temperatures exceeded the state rearing and migration temperature standard of 18˚C at six monitoring sites in 2013 for a total of 81 days, at more and farther upstream sites than in the previous three years, and for more days than in any year since 2007. Temperatures approached, but never exceeded, the 24˚C lethal threshold for salmon and steelhead in 2013, for the fourth year in a row. Regression of 1995-2013 temperature and flow data identified 56-65 cfs as the range of minimum flows necessary to meet the 18° temperature standard at FS Road 6360, slightly lower than the 66-78 cfs range predicted by 2008 and 2010-2012 models but well above the 33 cfs state water right. Temperature results from 2013 suggest that despite significant gains made in streamflow restoration and temperature reductions over the past ten years, at current flow protection levels temperatures are likely to continue to exceed fish requirements and state standards for rearing and migration in years characterized by lower flows. Continued development of creative solutions to allocation of Whychus Creek streamflow in low-water years is needed to guarantee conditions that will support the recovery of re-introduced native fish populations. Additionally, a better understanding of habitat use in Whychus and temperature tolerances of resident and anadromous O. mykiss and Chinook salmon at different life stages would allow restoration partners to refine strategies to provide sufficient temperature conditions. These results contribute to an improved understanding of temperature and flow on Whychus Creek that will allow restoration partners to better plan future watershed restoration efforts.

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. In 1998, 2002, 2004, and 2010, Whychus Creek was 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). The 2012 Integrated Report is pending (ODEQ 2014).

UDWC began monitoring temperature on Whychus Creek in 1995. In 1999 DRC streamflow 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 streamflow 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.

Water temperature affects the growth and survival of aquatic organisms. Temperature naturally fluctuates on both 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 streamflow 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 have been established 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 (7DMAX) temperatures are not to exceed 18˚C. The 2002 303d list also identified Whychus Creek as not meeting the 12.8˚C state temperature standard for salmon and steelhead spawning. Neither the 2004 nor the 2010 303(d) lists 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 may resume, and the spawning temperature standard become relevant, as salmon and steelhead reintroduced in 2007 begin to return to the creek. Steelhead spawning season has yet to be identified for Whychus Creek. This report references the January 1 – May 15 spawning season identified for the Lower Deschutes sub-basin. The State of Oregon 1992-1994 Water Quality Standards Review (ODEQ 1995) identified 24˚C as the lethal temperature threshold for salmon and trout.

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.

The 2004 and 2010 ODEQ 303(d) lists classify Whychus Creek as having insufficient data for assessment for dissolved oxygen and pH. UDWC analyses of data collected from 2006 to 2008 indicated that

Whychus Creek met state dissolved oxygen standards for steelhead and salmon 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 can 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 an appropriate 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 streamflow restoration. Accordingly this parameter was also discontinued as of 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 Streamflow Restoration Targets* (Jones 2010).

The streamflow 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 fish populations by reducing warming rates, reconnecting the creek to floodplains and groundwater, and increasing the extent of riparian shading. DRC and restoration partners identified a streamflow target for Whychus Creek according to state 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 streamflow to fish habitat availability (Thompson 1972), however minimum flows identified may not be sufficient to create suitable conditions for fish or meet state temperature standards. The DRC streamflow 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.

Whychus Creek Water Quality Status, Temperature Trends, and Streamflow Restoration Targets (Jones 2010) presented baseline information on the status of temperature, dissolved oxygen, and pH in Whychus Creek, demonstrated temperature response to increased flows from streamflow restoration in three downstream reaches, and developed a regression model describing the relationship between temperature and flow to identify streamflow restoration targets. Subsequent versions of this report incorporated 2009-2012 data to update 2006-2008 analyses (Mork 2012a, Mork 2012b, Mork 2013). We present revised analyses including 2000-2013 temperature and flow data to evaluate the 2013 status of temperature in Whychus Creek in relation to state standards for salmonid spawning, rearing and migration, quantify changes in temperature in reaches with restored flows, and refine flow targets projected to produce temperatures that meet state standards.

Table 1. 2010 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	12° 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 2011

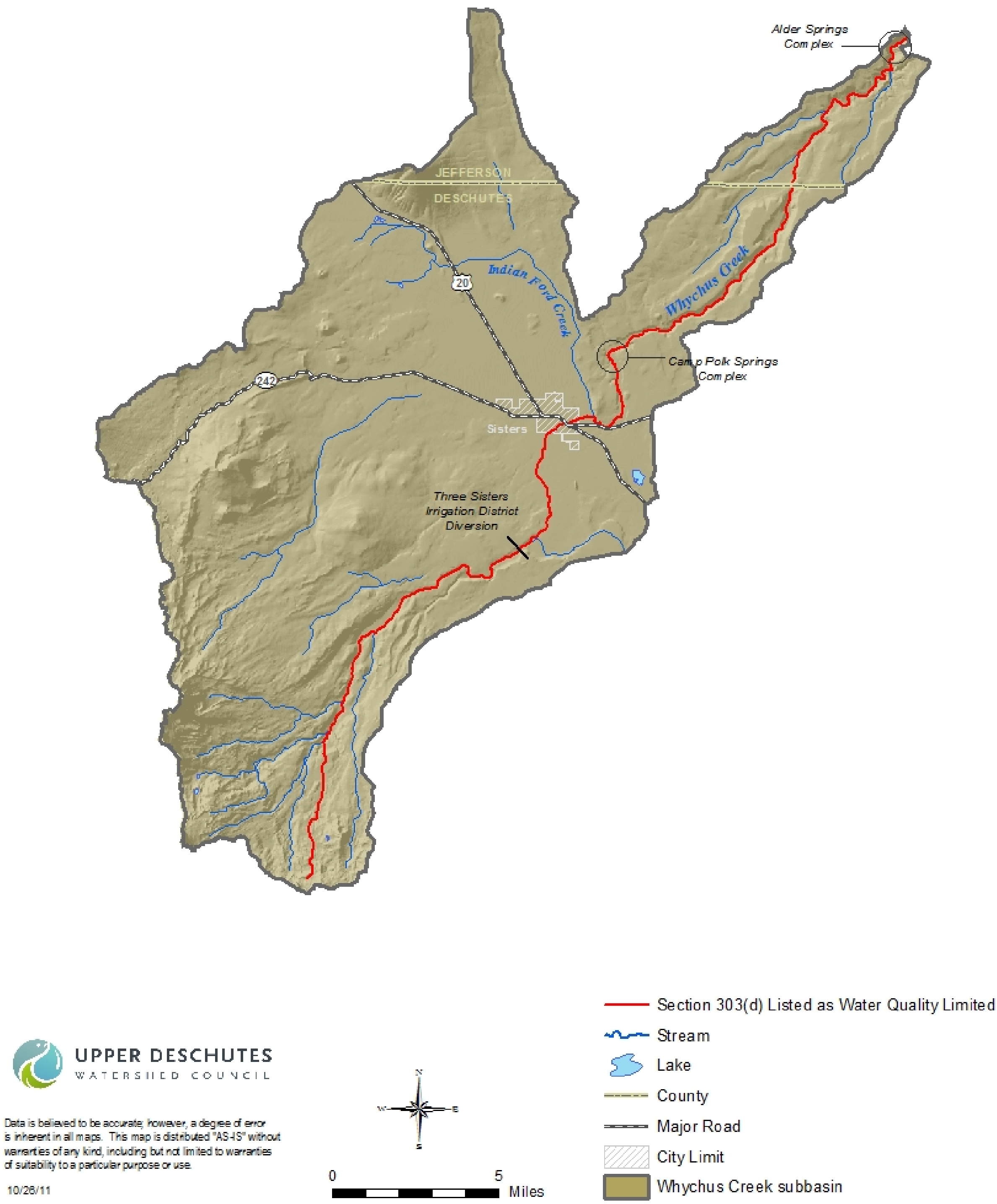


Figure 1.
Whychus Creek is listed as Water Quality Limited from river mile (RM) 0.0 to RM 40.3 under ODEQ's 2002, 2004, and 2010 303(d) lists. (ODEQ 2011)

Methods

Data collection

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 channel of Whychus Creek at Rimrock Ranch. The planned restoration will divert the creek from the existing channel into the historic meadow channel, and the UDWC monitoring station historically located on the existing channel will no longer be creekside. 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 these two locations.

We obtained average daily streamflow (QD) data for Whychus Creek from Oregon Water Resources Department (OWRD) gage 14076050 at the City of Sisters (OWRD 2013). This gage is located downstream from the Three Sisters Irrigation District diversion and other major irrigation diversions. We used data collected at this gage from 2000 to 2013 in this report, including some data considered by OWRD to be provisional and subject to change.

Temperature status

We evaluated 2013 seven day moving average maximum daily temperatures (7DMAX) in relation to the 18˚C state temperature standard for salmonid rearing and migration and the 12.8˚C state standard for salmonid spawning according to methods described in the ODEQ *Assessment Methodology for Oregon's 2004/2006 Integrated Report on Water Quality Status* (ODEQ 2006). At monitoring sites where July 7DMAX temperatures exceeded the18˚C standard, we compared temperatures to average daily and monthly median flows at Sisters City Park in relation to ODFW water rights and DRC streamflow restoration targets. We also compared the status of Whychus Creek temperatures in 2013 to 2000-2012 results.

We calculated the average longitudinal rate of temperature change for Whychus Creek on the hottest water day of 2012 from the farthest upstream site at RM 38 (WC 038.00) to the farthest downstream site at RM 0.25 (WC 000.25) as well as the average rate of change for each reach between monitoring sites along the creek by dividing the difference in temperature between these two sites by the distance between the two sites. We defined the hottest water day as the day with the single hottest seven day moving average maximum temperature (7DMAX). In 2013, because the hottest 7DMAX occurred on two consecutive days, we identified the day with the highest cumulative temperature across all sites as the hottest water day. We compared the rate of change for each reach to the average rate of change from the upstream-most to the downstream-most site. Higher than average longitudinal changes in temperature identify reaches in which the rate of warming increased, allowing restoration partners to prioritize these areas for evaluation and restoration planning. Lower than average longitudinal changes in temperature highlight reaches where cooling occurred and which may warrant prioritization for additional conservation measures.

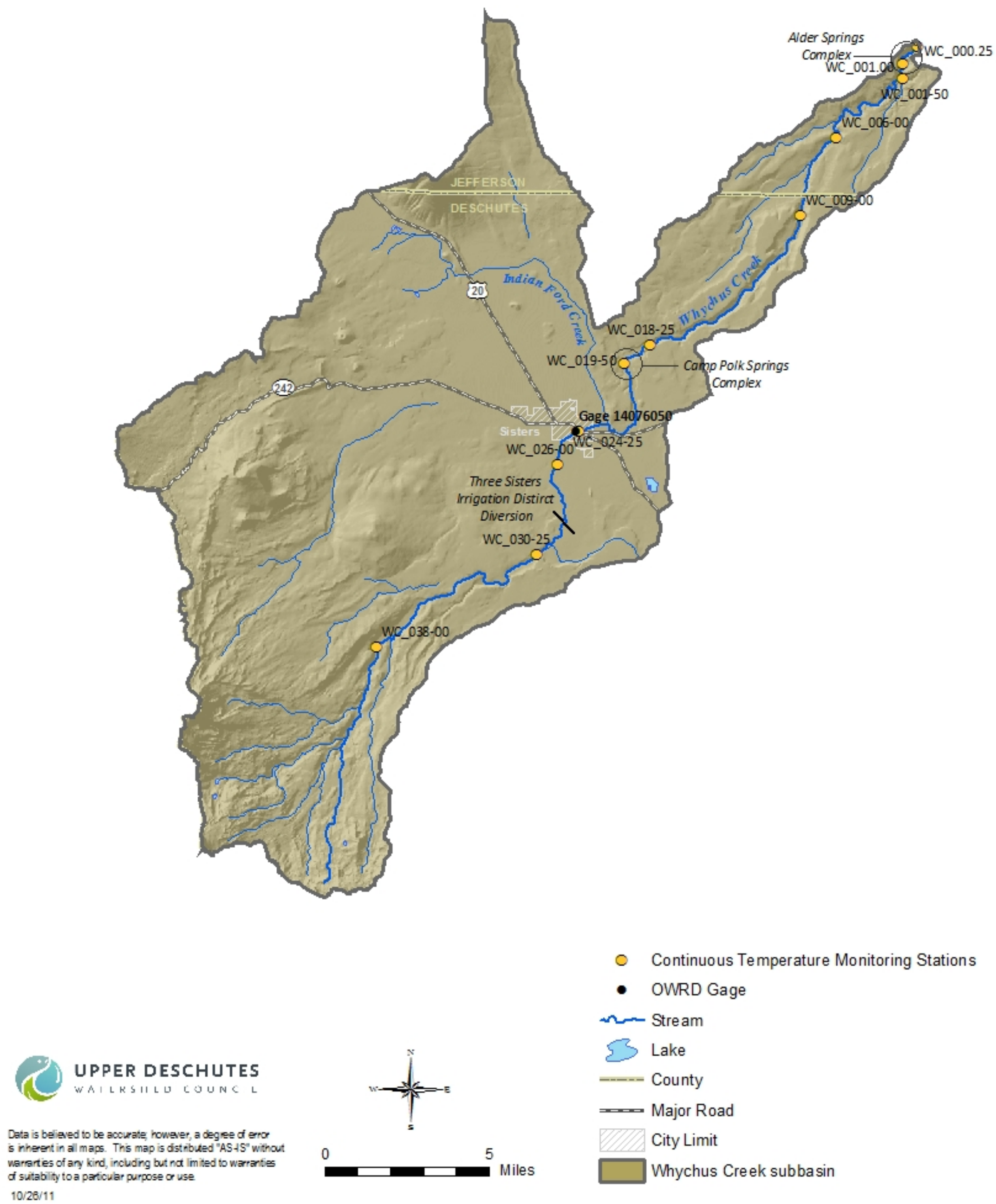


Figure 2.
Continuous temperature monitoring stations monitored in 2013, and OWRD Gage 14076050, on Whychus Creek.

Target Streamflow

We added 2013 data to the 2000-2012 dataset to perform a temperature-streamflow regression that refines the target streamflow projected to result in temperatures at or below the 18˚C state standard. We used July 7DMAX temperature data for each year included in the analysis from two monitoring stations, WC 024.25 and WC 006.00, with streamflow data from the OWRD gage at Sisters, to identify the streamflow required at each of these sites to achieve a 7DMAX temperature of 18˚C. Temperature data from WC 024.25 represent stream conditions immediately below major irrigation diversions; data from WC 006.00 represent the historical 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 streamflow restoration.

We restricted data included in the regression to one month of the year to reduce the effect of inter-annual seasonal variation in the analysis (Helsel and Hirsch 2002) and identified July as the month during which the hottest water day has historically occurred most frequently (UDWC 2003; UDWC 2008c). Daily streamflow data for all July days from 2000-2013 were collected at OWRD gage 14076050 at Sisters and downloaded from the OWRD Near Real Time Hydrographic Data website as average daily flow (QD; OWRD 2014).

To estimate temperatures at corresponding flows for the two locations we performed a regression of temperature and flow data. The resulting equations accurately represent the relationship between flow and temperature and can be used to calculate temperature values for the specified locations, within the evaluated time period, and within the range of flows observed. We paired 7DMAX temperature with the corresponding natural log of the average daily flow (LnQD) for each July day included in the analysis, then ranked flow values, assigned all July temperature records to their corresponding flow value, and calculated the mean of all 2000-2013 July 7DMAX temperatures observed at each flow level (Mork and Houston 2013). The seven day moving average maximum temperature for a given day is the average of the maximum temperature for that day, the three days prior, and the three days following; we paired the 7DMAX for a given day with the flow for the same day to best match the 7DMAX temperature to flow conditions on both the first and seventh days represented by the 7DMAX temperature. While this approach does not reflect flows corresponding to maximum daily temperatures on the fifth, sixth, or seventh days included in the 7DMAX temperature, we selected the flow corresponding to the 7DMAX for the same date as potentially more representative of the flow on both the first and seventh days included in the 7DMAX than if we used the flow from the seventh day to correspond to the 7DMAX temperature and thereby to represent the flow corresponding to the temperature from the first day of the 7DMAX.

We plotted flow versus mean temperature and used an ANOVA in R open source statistical software to determine the highest polynomial term that statistically improved the regression model (linear, quadratic, cubic, or quartic) on the basis of the R^2 value and p-value associated with each model. R^2 represents the proportion of the variation in mean 7DMAX temperatures that is explained by streamflow (Ln QD). As the fit of the regression to the data improves, the R^2 value increases toward a maximum 100%. Using the resulting regression equation for each location, we calculated the predicted temperature and 95% confidence interval for all flows within the observed range (**Appendix B**). We calculated the 95% confidence interval (CI) as:

$$Y \pm Y^{(Z_{1-\alpha/2} S(x) / \sqrt{N})}$$

where $Z_{1-\alpha/2} = Z_{1-0.05/2} = Z_{0.475} = 1.9$ (NIST 2011)

We compared the resulting 2000-2013 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 33 cfs is the DRC streamflow restoration target, and because available Heat Source scenarios assume 33 cfs at WC 024.25 and 62 cfs at WC 006.00, we compared 2000-2013 temperature calculations to the Heat Source estimates for these flows.

Results

Temperature status

Although seven-day moving average maximum (7DMAX) temperatures have decreased overall since 2000, temperatures increased in 2012 and 2013 (Figure 3, Figure 4), with 2013 temperatures in Whychus Creek the highest observed since 2007. Temperatures at Sisters City Park, three miles downstream of the most significant irrigation diversion on the creek, exceeded the 18˚C standard in 2013 for the first time since 2007, over 20 days at flows of 19 to 36 cfs, following at least four years of temperatures remaining below 18˚C at this site. Temperatures at this site exceeded the 18˚C standard in every year for which data were available prior to 2006. Temperatures exceeded the 18˚C state standard for steelhead and salmon rearing and migration at six sites total in 2013, at more and farther upstream sites than in the previous three years, for 20 days along 18 miles of creek and for 54 days along 12 miles. Data were not available in 2013 for WC 001.50, where temperatures have exceeded 18˚C in every year for which data are available since 2008.

At Road 6360 (WC 006.00), historically the most impaired site for which temperature data are available, 7DMAX temperatures exceeded the 18˚C standard at flows of 18 to 117 cfs for 81 days in 2013, the most days in excess of 18˚C at any location in the creek since 2007, and the third highest total number of days above 18˚C of the nine years for which data are available for this site (Figure 4). Despite high temperatures for extended periods in 2013, lethal temperatures (24˚C) were not recorded at any site on Whychus Creek in 2013, for the fourth year in a row, although they hovered above 23˚C for 6 days in July.

Median July temperature trends at Sisters City Park and Road 6360 closely tracked July median flow for all years for which data were available (Figure 4, Figure 5). July 2013 median temperatures corresponded to the lowest July median flow since 2007; 2007 was characterized by the highest temperatures and lowest flows since 2005. Years characterized by the lowest temperatures observed in Whychus Creek also experienced the highest July median flows since stream flow restoration began.

Continuous temperature data were available for the 2013 spawning season for 35-43 days during the spawning season, varying by site (Figure 3). Temperatures recorded for these dates exceeded the 12.8˚C spawning habitat requirement and potential state standard at six sites over 8 to 23 days from the first date in April for which data were available to May 15, representing 19%-61% of spawning season days for which data were available. This represents the highest number of spawning season days for which data are available characterized by temperatures that exceeded spawning criteria since 2007.

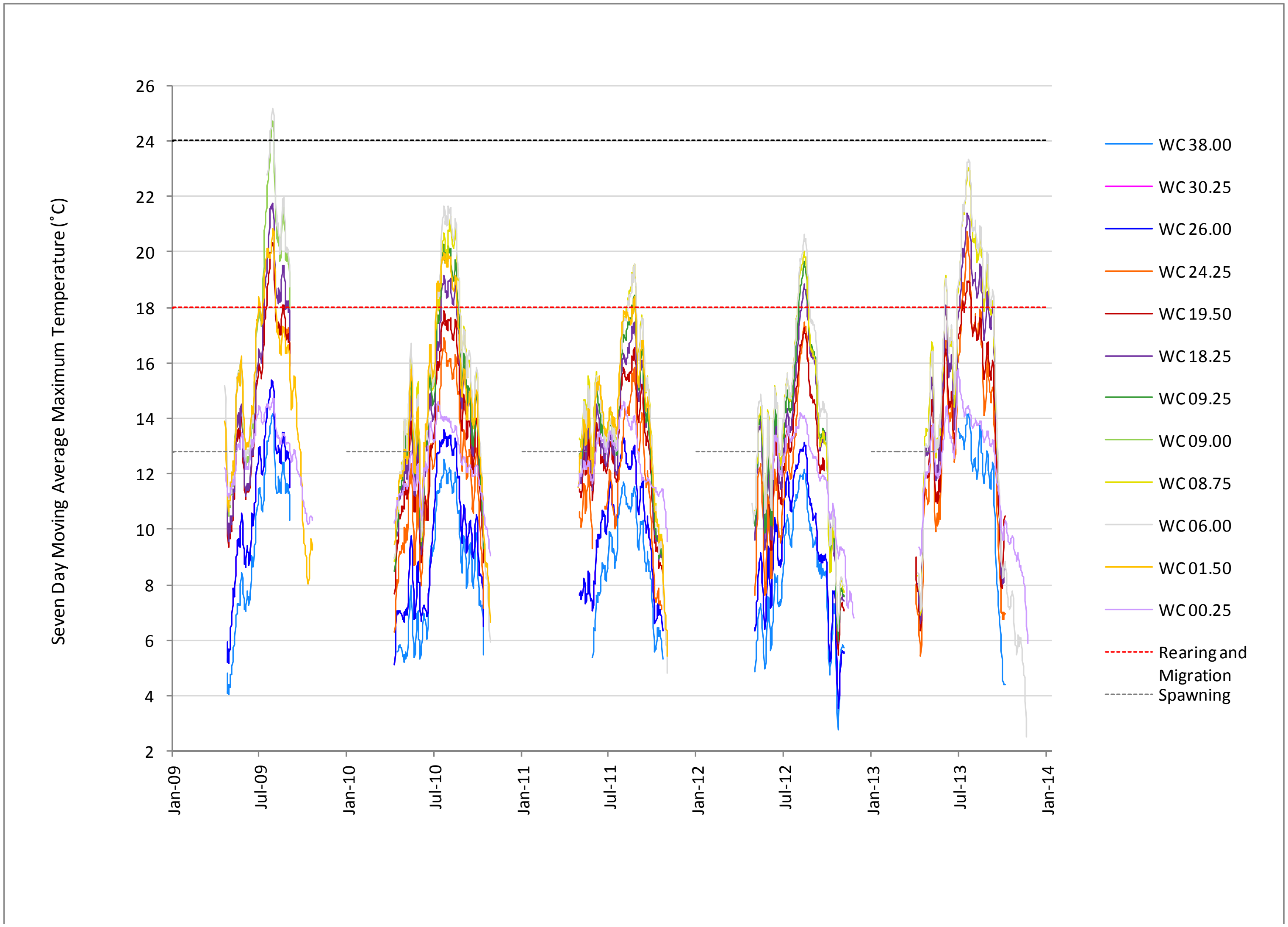
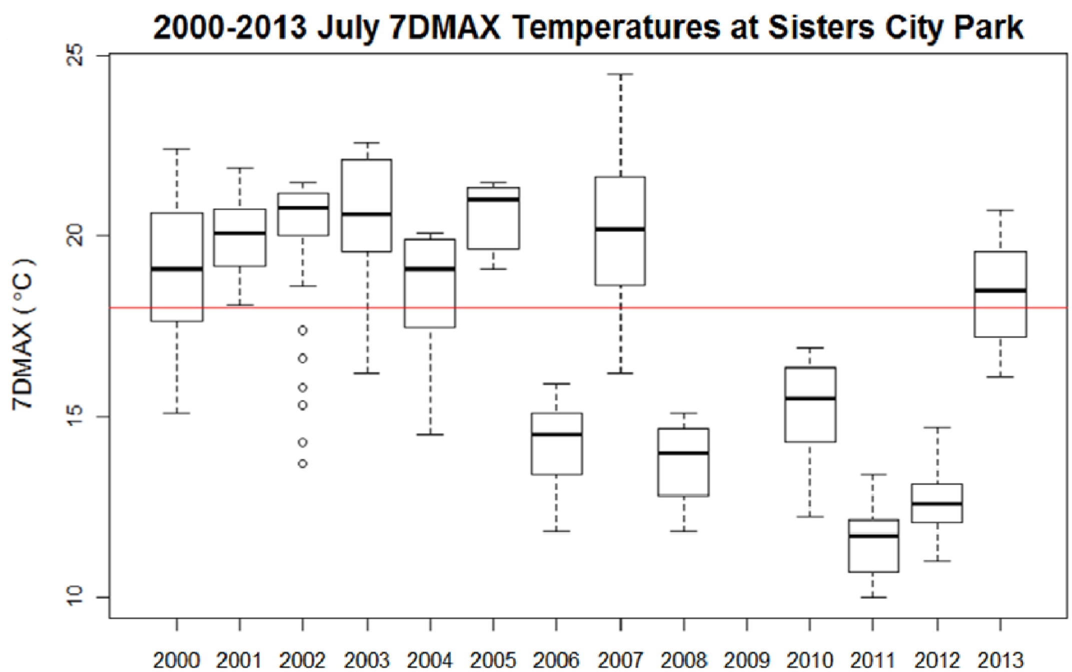


Figure 3.
Temperatures exceeded the 18˚C state standard for salmon and trout rearing and migration and the January 1- May 15 12.8˚C potential spawning standard at six monitoring locations in 2013, more than in the previous three years. Temperatures approached the 24˚C lethal threshold in 2013, but have not met or exceeded this threshold at any monitoring location along Whychus Creek since 2009.

a



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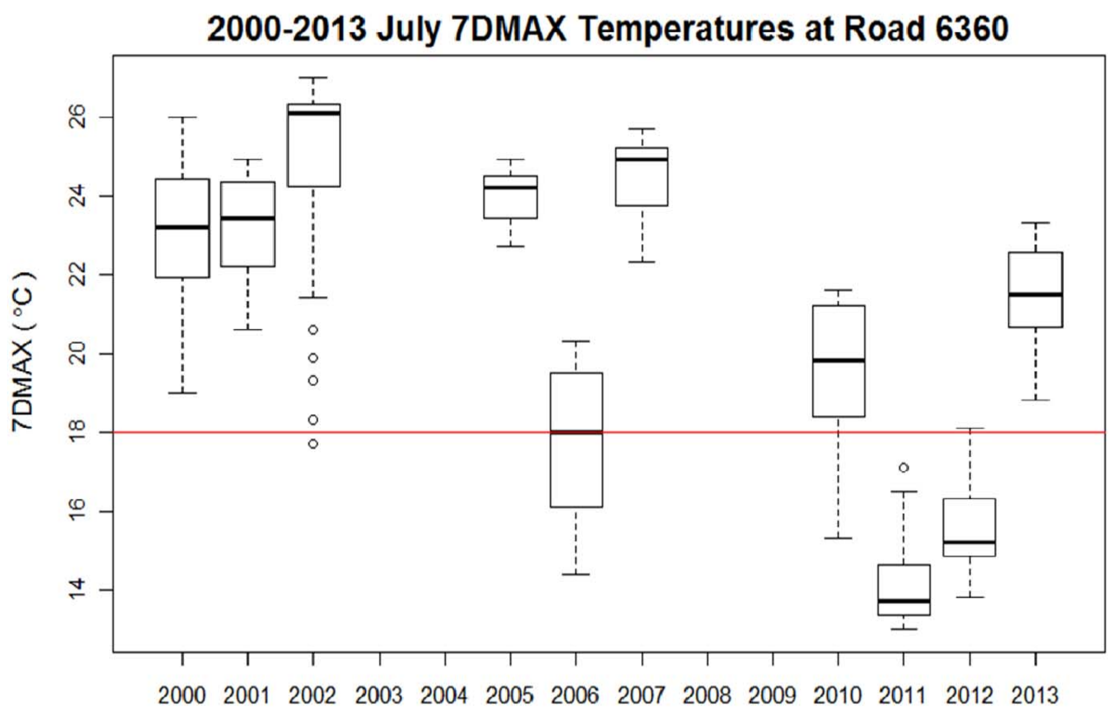


Figure 4.
July mean 7DMAX temperatures at a) Sisters City Park (WC 024.25) and b) Road 6360 (WC 006.00) decreased substantially between 2001 and 2011 but have increased since. Despite temperatures at Sisters City Park increasing in 2012 and 2013, they have remained below the 18˚C standard, exceeding it in only one year for which data are available since 2006; no data are available for this site for 2009. Temperature trends at Road 6360 track temperatures at Sisters City Park, with commensurate increases in 2012 and 2013 over 2011, but continue to exceed the 18˚C standard despite falling dramatically since 2001

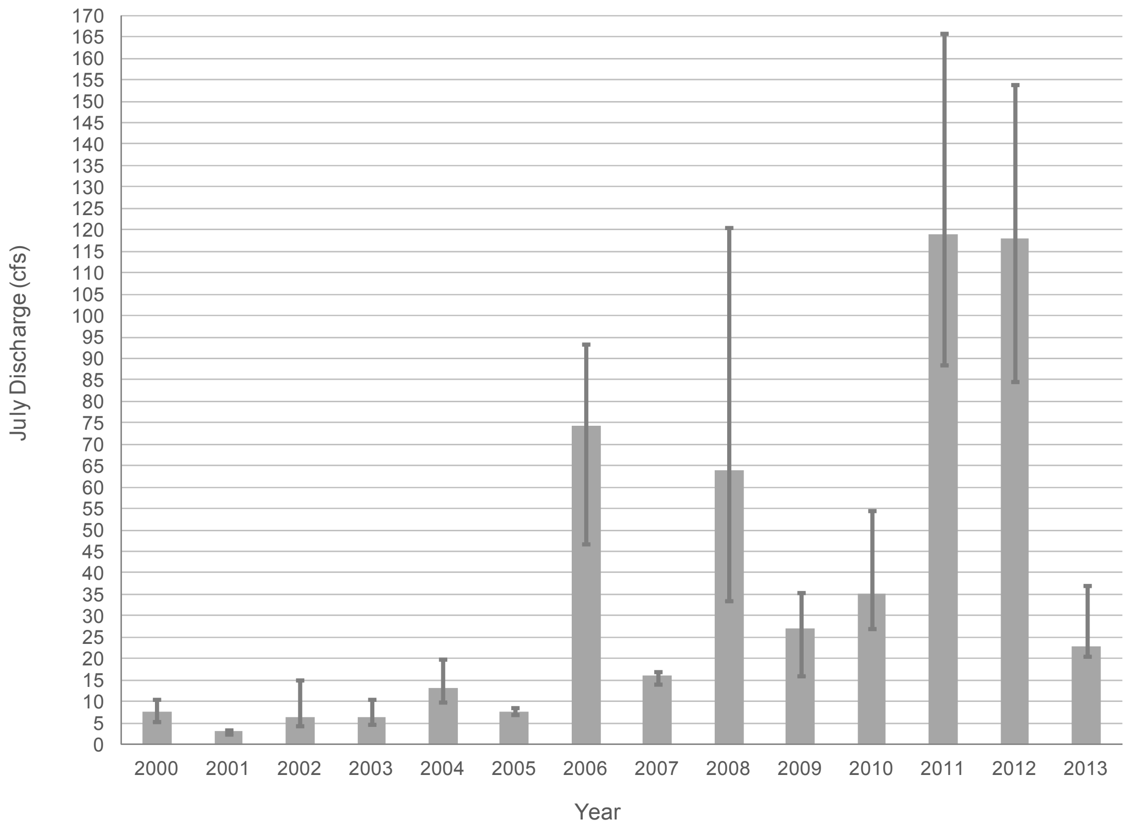


Figure 5. July median streamflow from 2000 to 2013 corresponded closely to median temperatures at Sisters City Park and Road 6360.

The hottest water day in 2013 occurred on July 22nd. Temperatures on this day were higher than hottest day temperatures for 2012, 2011, 2010 and 2008, but lower than hottest day temperatures in 2009 and 2007 at most sites. 2013 hottest day temperatures at Road 1514 (WC 038.00) and above Road 4606 (WC 026.00) were the highest of any year from 2007 to 2013 (Figure 6). Data were not available for 2013 from WC 030.25.

The average longitudinal rate of change in 2013 was 0.01˚C per mile, lower than the average rate of change for any year for which this statistic has been calculated, but close to the 2009 average rate of 0.02˚C per mile (Figure 7). Importantly, the relatively lower average longitudinal rates of change for 2013 and 2009 reflect markedly higher temperatures than in other years at the upstream-most sites, above significant irrigation diversions, with temperatures at the downstream-most site within 0.07˚C for all years between 2009 and 2013. This result suggests that higher temperatures along Whychus Creek in 2013 may be attributed in part to the initially high temperature of streamflow upstream of irrigation diversions or any other direct anthropogenic alterations to the hydrograph.

Above-average warming and cooling occurred at the same sites as in previous years, but were more extreme in 2013 than in previous years. Whereas the greatest increase over the average rate of change in 2007, 2008, 2010 and 2011 occurred between WC 026.00 and WC 024.25, downstream of the TSID diversion and upstream of Sisters, in 2013, as in 2012, the greatest increase over the average rate of change occurred along the reconstructed channel at Camp Polk, from WC 019.50 to WC 018.25 (data from WC 024.25 are not available for 2009). However, cooling between Sisters City Park and the upstream Camp Polk site at WC 019.50 was first observed in 2012 and was more pronounced in 2013, resulting in a lower WC 019.50 temperature which contributed to a greater difference in temperature, and therefore a higher rate of warming, between the upstream, WC 019.50 site and the downstream, WC 018.25 site. Cooling between Sisters City Park and WC 019.50 suggests an increased cooling effect of spring water inflow upstream of RM 19 (Watershed Sciences 2008) and may indicate increased flow from these springs, potentially associated with elevation of groundwater and floodplain reconnection following channel and floodplain restoration in Camp Polk Meadow. Increased warming in the Camp Polk Meadow reach may also reflect increased solar insolation along the new Camp Polk channel, in the absence of the mature riparian canopy that shaded the straightened channel in previous years, and a longer residence time for stream flow slowed by channel meanders. The 2013 rate of warming between WC 026.00 and WC 024.25 was second only to that along the Camp Polk channel. The rate of warming through Rimrock Ranch, historically also high, was unavailable for 2013 due to the datalogger for the downstream site missing at the conclusion of the 2013 monitoring season. The -1.5˚C rate of temperature change from Road 6360 to Alder Springs was substantially lower than the average rate of change, reflecting the cooling effect of springs complex flows, but less dramatically so than in 2011 (-3.4˚C) and previous years.

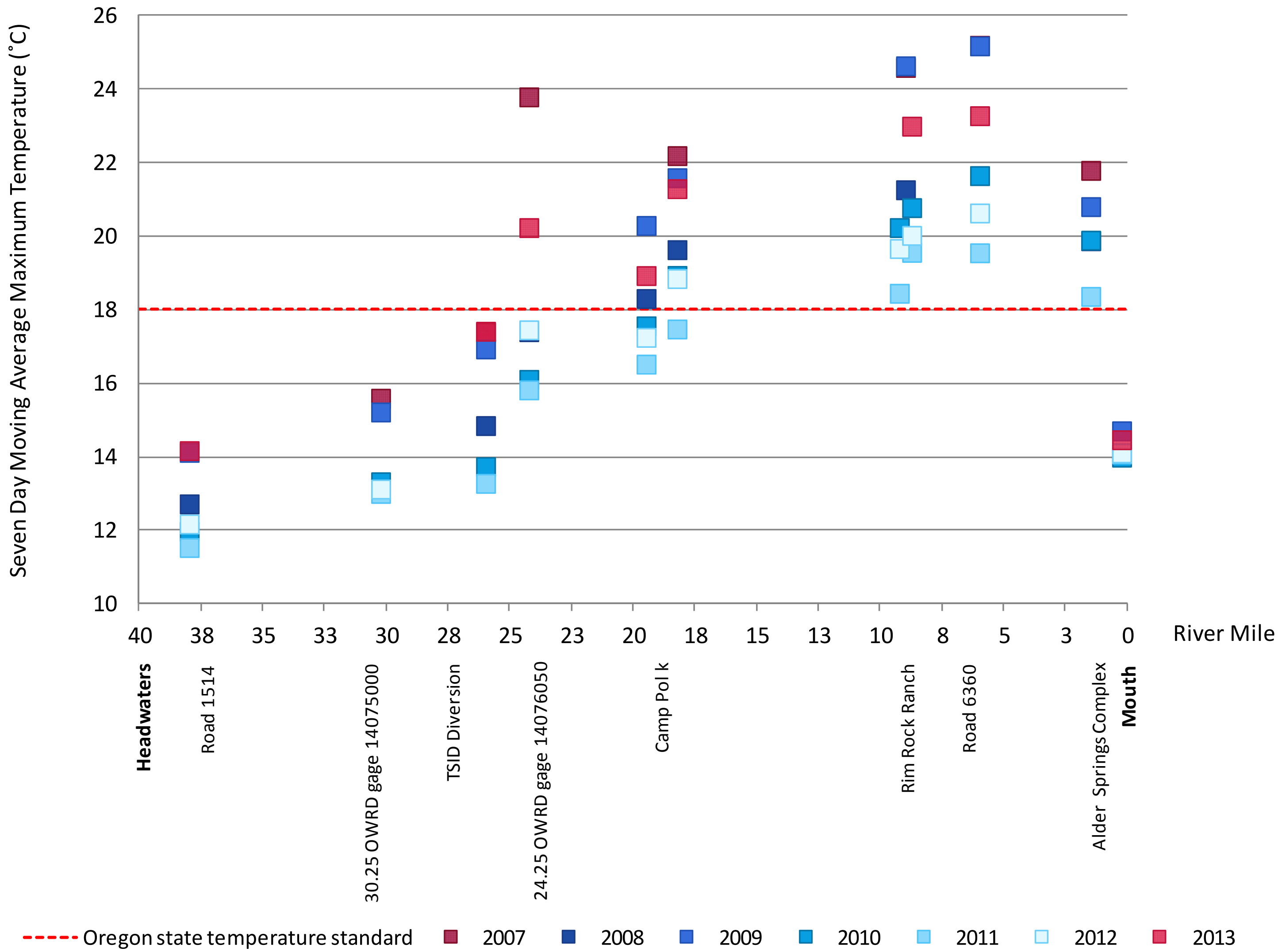


Figure 6.
The longitudinal temperature profile of Whychus Creek on the hottest water days from 2007 through 2013 shows warming from the headwaters to Alder Springs, where springs complex flows cool stream temperatures.

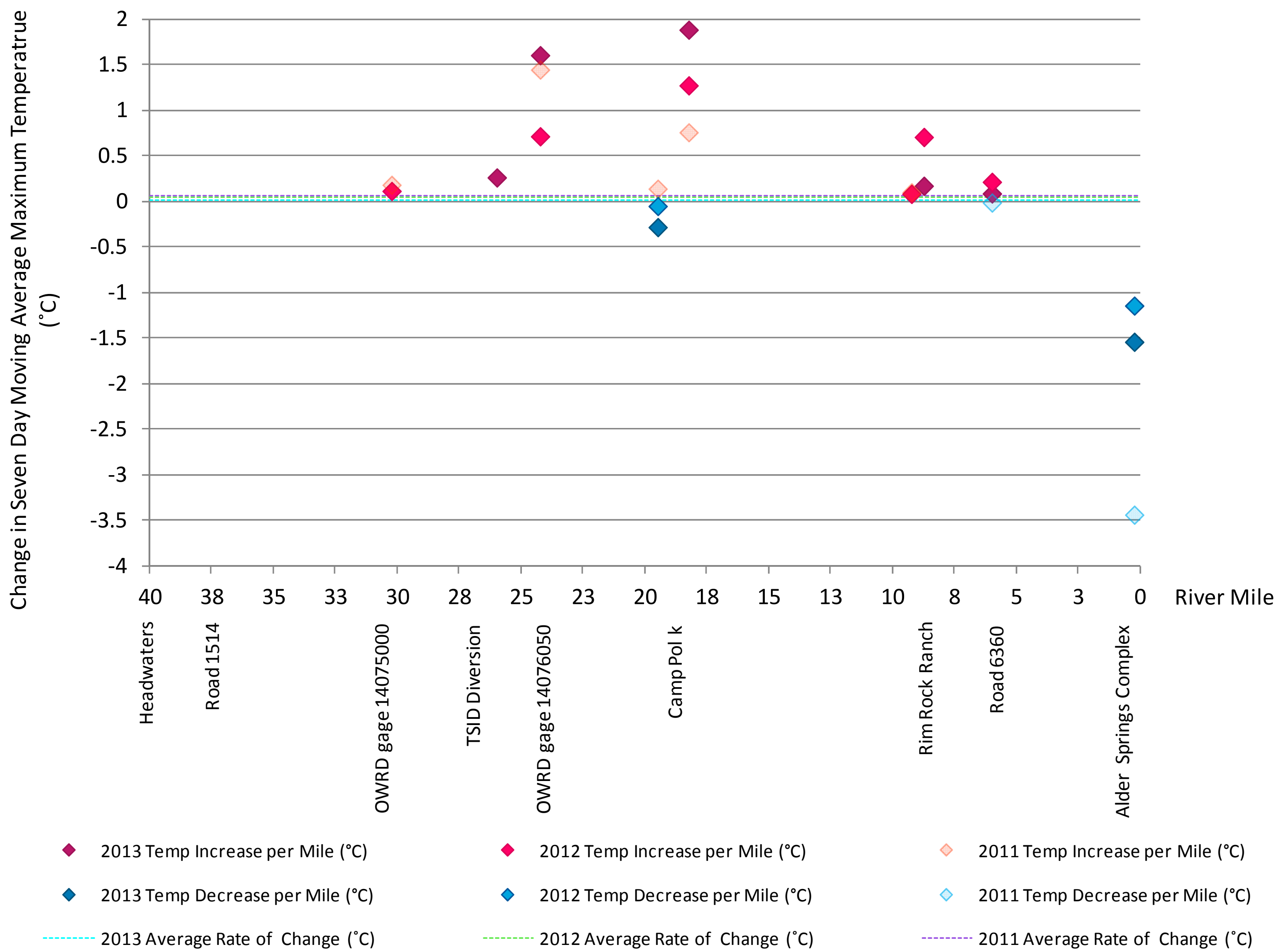


Figure 7.
Longitudinal average rate of temperature change in Whychus Creek in 2011-2013. Higher than average longitudinal changes in temperature identify reaches in which the rate of warming increased and suggest prioritization of these areas for research and restoration planning. Lower than average longitudinal changes in temperature highlight reaches where cooling occurred, indicating an opportunity to preserve optimal conditions through additional conservation measures.

Target streamflow

The quartic model was statistically better than lower-order models for both WC 24.25 and WC 006.00 data (Table 3, Figure 8). Temperatures calculated from the WC 024.25 regression model suggest that 20 cfs (3.0 LnQD) was the minimum streamflow resulting in a mean 7DMAX temperature at or below 18°C (± 1.6°C) given temperatures observed from July 2000-2013 at this location. The existing 33 cfs restoration target resulted in a mean 7DMAX temperature of 15.9°C ± 1.5°C (Appendix B), well below the 18°C standard. 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 2000-2013 estimate for Sisters City Park is close to 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 2000-2013 model estimate is within the range of estimates from earlier temperature-flow relationships which predicted temperatures meeting the state standard at flows of 19-21 cfs (Jones 2010, Mork 2012a, Mork 2012b, Mork 2013).

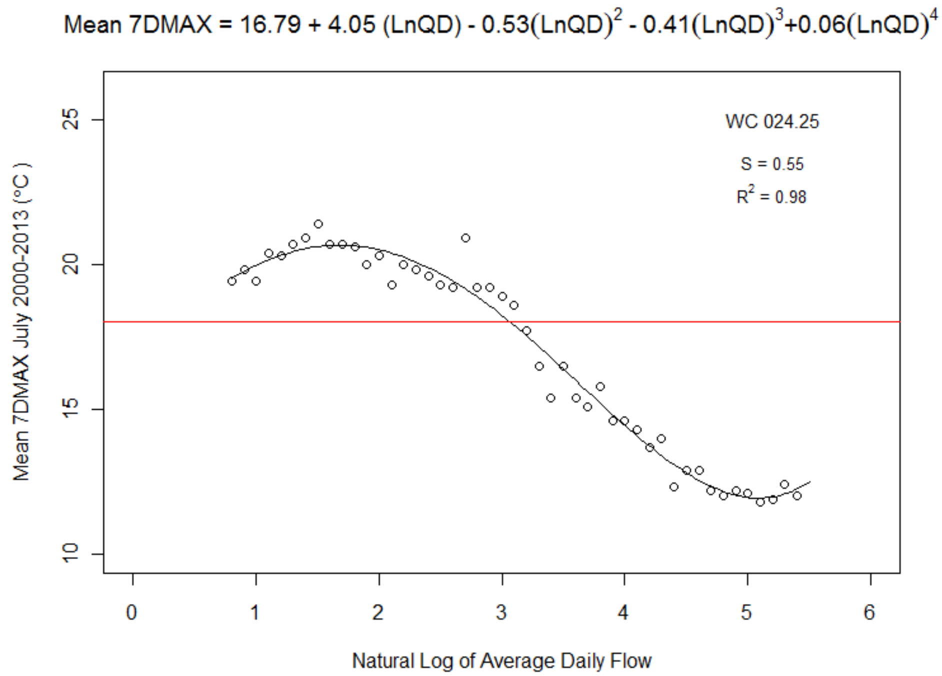
Although the quartic regression model was strongly significantly better than cubic and lower-order models, we evaluated temperatures calculated from both the cubic and quartic models at given flows for WC 006.00 (**Appendix B**). At flows between 10 cfs (2.3 LnQD), temperature estimates from the two equations were within tenths of a degree. Temperatures calculated from the quartic equation were consistently higher (more conservative) between 10 and 32 cfs than temperatures calculated from the cubic equation. Above 32 cfs temperatures calculated from the cubic equation were higher than temperatures calculated from the quartic equation, with the difference in temperatures increasing through the range of flows from tenths of degrees at 32 cfs to a full degree at 89 cfs, and with differences between models diminishing above 160 cfs. The quartic model predicted 18°C at 56 cfs, lower than any previous model with the exception of the 2012 cubic model which predicted 18°C at 53 cfs; the cubic model predicted 18°C at 65 cfs, also lower than any previous model other than the 2012 cubic model. Confidence intervals for the 2013 quartic model were slightly lower than for the cubic model.

Cubic and quartic regression models for temperature-flow relationships at Road 6360 (WC 006.00) derived from 2000-2013 temperature and flow data predict a range of 56-65 cfs (4.0-4.15 LnQD) to be the minimum streamflow that will achieve a mean 7DMAX temperature of 18.0°C ± 1.7-1.8°C. Under these models the target streamflow of 33 cfs below Indian Ford Creek is projected to produce a mean 7DMAX temperature of 20.5 (quartic) - 20.6 (cubic)°C ± 1.7-1.8°C at Road 6360, above the 18°C state standard but below the 24°C lethal temperature threshold. The 2000-2012 quartic regression model predicts 17.5 (quartic) to 18.2 (cubic)°C ± 1.7-1.8°C at 62 cfs, slightly lower than the Heat Source model estimate of 18.5°C ± 1°C at 62 cfs at Road 6360.

Table 3. A quartic regression model provided the best fit to 2000-2013 temperature-flow data for both WC 024.25 and WC 006.00 data. Temperatures calculated using the corresponding regression equations are expected to be the most accurate of the possible regression models.

Regression	Equation	df	S	R^2	p-value
	WC 24.25 (n=47)				
Linear	24.11-2.31(LnQD)	45	1.15	0.88	
Quadratic	$20.88+0.27(LnQD)-0.42(LnQD)^2$	44	0.93	0.92	0.00
Cubic	$13.68+9.67(LnQD)-3.84(LnQD)^2+0.37(LnQD)^3$	43	0.56	0.97	0.00
Quartic	$\mathbf{16.79+4.05(LnQD)-0.53(LnQD)^2-0.41(LnQD)^3+0.06(LnQD)^4}$	**42**	**0.55**	**0.97**	0.08
	WC 06.00 (n=46)				
Linear	28.32-2.47(LnQD)	44	1.47	0.84	
Quadratic	$22.78+2.01(LnQD)-0.74(LnQD)^2$	43	0.89	0.94	0.00
Cubic	$17.08+9.55(LnQD)-3.53(LnQD)^2+0.31(LnQD)^3$	42	0.68	0.97	0.00
Quartic	$\mathbf{22.74-0.83(LnQD)+2.68(LnQD)^2-1.17(LnQD)^3+0.12(LnQD)^4}$	**41**	**0.63**	**0.97**	0.01

a



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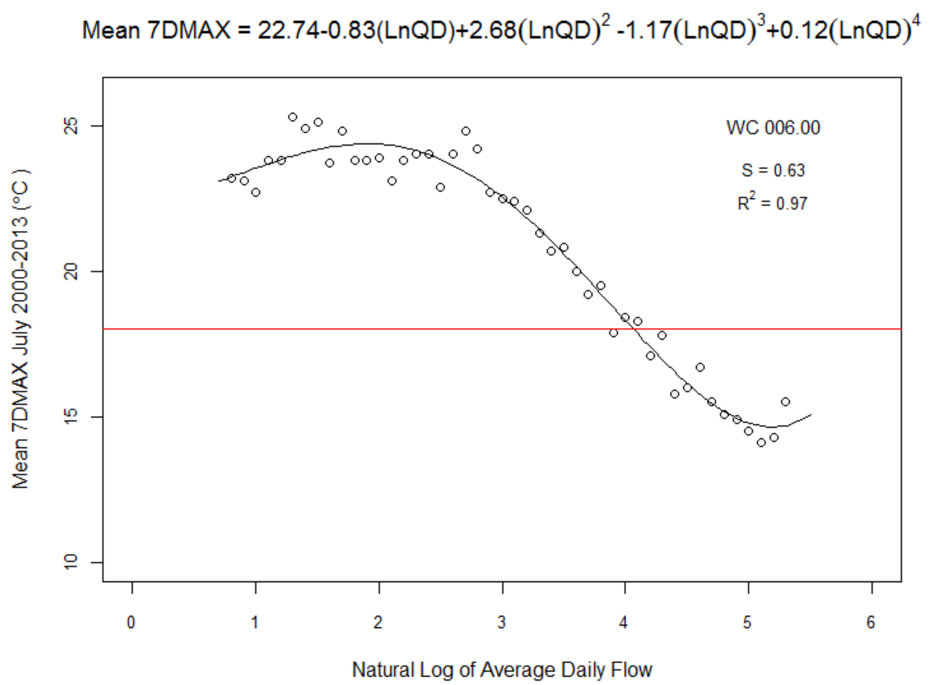

Figure 8.

Regression models fitted to temperature-flow data from July 2000-2013 describe the relationship between temperature and flow observed at a) Sisters City Park (WC 024.25), and b) Road 6360 (WC 006.00), fitted with quartic regression model trendlines. We used the corresponding regression equations to calculate temperature at a given flow.

Discussion

Temperature status

Temperatures in excess of 18˚C over a prolonged duration along the length of Whychus Creek in 2013 support the ODEQ 2010 303(d) Category 5 listing of Whychus Creek as water quality limited (ODEQ 2011). Because the state standard is biologically based, we can infer that temperature conditions above 18˚C compromised habitat suitability for rearing and migrating trout and salmon, *at a minimum in some locations,* along Whychus Creek in 2013. Seven day moving average maximum temperatures also indicate marginal conditions for spawning trout and salmon. 7DMAX temperatures did not exceed the 24˚C lethal threshold in 2013.

Temperature data from 2008 through 2013 show temperatures in Whychus Creek closely tracking flow conditions. From 2008 through 2012, at July median flows of 35-119, 7DMAX temperatures exceeded the state standard for many fewer days and at fewer sites than in previous years, with the highest temperatures occurring later in the summer. The cooling trend observed from 2007 to 2012 was interrupted by higher 2013 temperatures that exceeded the state standard at more sites and over a longer duration than any observed since 2007. Although 7DMAX temperatures greater than 18˚C were observed at flows as high as 117 cfs in July 2013, median stream flow was only 23 cfs; the previous lowest July median flow for which temperature data are available, 16 cfs in 2007, corresponded to the next highest July temperatures, and so on for median July temperatures and flow back to 2000.

Stream flow restoration effectiveness

Regression of temperature and flow data as well as comparison of median monthly temperature and stream flow data and mean 7DMAX temperatures for given flow levels show stream temperatures decreasing as flows increase. Stream flow restoration that has increased the minimum flow delivered instream has resulted in higher monthly median flows reflecting consistently higher average daily flows corresponding 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, Deschutes River Conservancy established a stream flow restoration target of 33 cfs for the entire length of the creek from headwaters to mouth. Regression results from Road 6360 (WC 006.00) 2000-2013 temperature and flow data indicate flows between 56 and 65 cfs are necessary to achieve stream temperatures at or below 18°C at this site. 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 native trout and salmon and support the conclusion of previous regression models and Heat Source model results (Watershed Sciences 2008). In addition, minimum flows that on average have resulted in 18°C will not be sufficient to meet that threshold in hotter years given the influence of air temperature on stream temperature, as highlighted by slightly higher stream temperatures in 2012 than in 2011 at nearly identical flows.

The degree to which high temperatures negatively impact native fish populations in Whychus Creek will depend on the spatial extent, duration, and severity of temperatures and on the actual biological requirements of local populations of redband, steelhead, and salmon. Although temperature requirements of local native populations are not well described, some recent studies suggest preliminary thresholds. Keefer et al. (2009) found that mid-Columbia steelhead use of small cool-water tributaries increased rapidly when mainstem Columbia River temperatures reached 19°C, and that use of thermal refuges resulted in other costs including lower survival to natal basins, higher rates of harvest, and greater unknown mortality. Goniea et al. (2006) showed migration rates of Chinook salmon slowed significantly at temperatures above 20°C while fish took refuge in tributaries averaging 2-7°C cooler than the mainstem Columbia. Studies are needed to document fish use of habitat in relation to temperature and cold-water inflows in Whychus Creek and elucidate what temperature conditions, within what spatial extent, are suitable to support healthy populations of native fish species. This information can in turn be used to identify interim targets for stream flow restoration that will better support basic biological functions in Whychus Creek and meet the restoration objective of restoring the stream conditions and processes necessary to support chinook salmon, redband trout, steelhead trout and bull trout (UDWC 2009). 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 and creating pools and cover for resident redband.

Conclusions

Temperatures in Whychus Creek appear to closely track July stream flow. Sustained high 2013 temperatures that exceeded the state temperature standard and biological rearing and migration requirements for up to 81 days are directly correlated to a low monthly median flow of 23 cfs for July; dramatic temperature reductions in 2012 and 2011 reflected historically high July monthly median flows of 119 and 118 cfs, respectively.

The temperature-flow relationship described by thirteen years of data indicates 56-65 cfs is the minimum range of flows at which stream temperature will meet 18°C. Given the relationship between stream flow and temperature observed in Whychus Creek, approaches to stream flow restoration that maximize mid-summer and in particular July flows up to 65 cfs will most reliably result in temperatures that create suitable stream conditions for rearing and migrating trout and salmon. Although 65 cfs may not currently be a feasible restoration target, these data nonetheless provide a benchmark for streamflow 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. A better understanding of habitat use in Whychus and temperature tolerances of resident and anadromous *O. mykiss* and Chinook salmon at different life stages would allow restoration partners to refine strategies to provide sufficient temperature conditions.

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, and contribute to an improved understanding of temperature and flow on Whychus Creek that will allow restoration partners to better plan future watershed restoration efforts with consideration for changing climate conditions.

References

Brett JR, Clar WC, Shelbourn JE. 1982. Experiments on the thermal requirements for growth and food conversion efficiency of juvenile chinook salmon. Canadian Technical report of fisheries and Aquatic Sciences 1127. Pacific Biological Station, Nanaimo, BC. 29 pp.

Goneia TM, Keefer ML, Bjornn TC, Peery CA, Bennett DH, Stuehrenberg LC. 2006. Behavioral thermoregulation and slowed migration by adult fall chinook salmon in response to high Columbia River water temperatures. Transactions of the American Fisheries Society, 135:2.

Jones L. 2010. Whychus Creek water quality status, temperature trends, and streamflow restoration targets p. 18-55 in Golden B, Houston R, Editors. 2009 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 134 p.

Helsel DR and Hirsch RM. 2002. Hydrologic Analysis and Interpretation, Chapter A3 *in* Statistical Methods in Water Resources, Techniques of Water Resources Investigations of the United States Geological Survey, Book 4. United States Geological Survey. http://water.usgs.gov/pubs/twri/twri4a3/

Keefer ML, Peery CA, High B. 2009. Behavioral thermoregulation and associated mortality trade-offs in migrating adult steelhead (Oncorhynchus mykiss): variability among sympatric populations. Canadian Journal of Fisheries and Aquatic Sciences 66:10.

Montgomery ML. 1953. Deschutes River Investigations, Central Region. River Basin Investigations, Oregon State Game Commission.

Mork LA. 2012a. Whychus Creek water quality status, temperature trends, and streamflow restoration targets p. 20-45 in Mork L, Houston R, Editors. 2010 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 72 p.

Mork LA. 2012b. Whychus Creek water quality status, temperature trends, and streamflow restoration targets p. 20-46 in Mork L, Houston R, Editors. 2011 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mork LA. 2013. Whychus Creek water quality status, temperature trends, and streamflow restoration targets p. 19-42 in Mork L, Houston R, Editors. 2012 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 110 p.

Nehlsen W. 1995. Historical salmon and steelhead runs of the Upper Deschutes River Basin and their environments. Portland General Electric, Portland, Oregon.

Nielson RS. 1950. Survey of the Columbia River and its tributaries. Part V. U.S. Fish and Wildlife Service Special Scientific Report: Fisheries No. 38.

NIST (National Institute of Standards and Technology). 2011. NIST/SEMATECH e-Handbook of Statistical Methods, http://www.itl.nist.gov/div898/handbook/. National Institute of Standards and Technology.

ODEQ (Oregon Department of Environmental Quality). 1995. State of Oregon 1992-1994 Water Quality Standards Review. Oregon Department of Environmental Quality. Portland, Oregon.

ODEQ (Oregon Department of Environmental Quality). 2006. Assessment Methodology for Oregon's 2004/2006 Integrated Report on Water Quality Status. Oregon Department of Environmental Quality. Portland, Oregon.

ODEQ (Oregon Department of Environmental Quality). 2009. Water Quality Standards: Beneficial Uses, Policies, and Criteria for Oregon. Oregon Administrative Rules, Chapter 340, Division 041. Oregon Department of Environmental Quality. Portland, Oregon.

ODEQ (Oregon Department of Environmental Quality). 2011. Oregon's water quality assessment 2010 integrated report. Oregon Department of Environmental Quality. Portland, Oregon. http://www.deq.state.or.us/wq/assessment/2010Report.htm Accessed online July 5, 2011.

ODEQ (Oregon Department of Environmental Quality). 2014. Oregon's 2012 integrated report. Oregon Department of Environmental Quality. Portland, Oregon. http://www.deq.state.or.us/wq/assessment/2012DataInfo.htm Accessed online February 17, 2014.

OWRD (Oregon Water Resources Department). 2003. Graphic chart showing the average natural August flow of the Deschutes River and tributaries for a 30-year average.

OWRD (Oregon Water Resources Department). 2014. Near real time hydrographic data. http://apps.wrd.state.or.us/apps/sw/hydro_near_real_time/display_hydro_graph.aspx?station_nbr=14076050. Accessed online January 30, 2014.

Thompson K. 1972. Determining stream flows for fish life. Presentation at Pacific Northwest River Basins Commission Stream Flow Workshop. March 15-16, 1972.

UDWC (Upper Deschutes Watershed Council). 2003. Temperatures of the Upper Deschutes and Little Deschutes Subbasins. Upper Deschutes Watershed Council. Bend, Oregon.

UDWC (Upper Deschutes Watershed Council). 2008a. Quality Assurance Project Plan; Water Quality Monitoring Program. Prepared by Jones L. Upper Deschutes Watershed Council. Bend, OR.

UDWC (Upper Deschutes Watershed Council). 2008b. Water Quality Monitoring Program Standard Operating Procedures - Field. Upper Deschutes Watershed Council. Bend, Oregon.

UDWC (Upper Deschutes Watershed Council). 2008c. Whychus Creek Temperature Summary 2005-2008. Upper Deschutes Watershed Council. Bend, Oregon.

UDWC (Upper Deschutes Watershed Council). 2009. UDWC Whychus Creek Restoration Monitoring Plan. Upper Deschutes Watershed Council. Bend, Oregon. 40 p.

USGS (United States Geological Survey). 2001. Ground-Water hydrology of the Upper Deschutes Basin, Oregon, Water Resources Investigation Report 00-4162. Portland, Oregon.

Watershed Sciences and MaxDepth Aquatics. 2008. Whychus Creek Stream Temperature Modeling: Various Flow Scenarios. *Addendum to* Deschutes River, Whychus Creek and Tumalo Creek Temperature Modeling. Deschutes River Conservancy. Bend, Oregon.

APPENDIX A Whychus Creek continuous temperature monitoring stations 1995-2013. TE = Temperature status, ST = Streamflow target.

Station ID	Description	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
WC 038.00	Road 1514				TE				TE				TE		TE	TE	TE	TE	TE	TE
WC 030.25	OWRD Gage 14075000					TE			TE	TE	TE	TE	TE	TE		TE	TE	TE	TE	TE
WC 026.00	Road 4606 Footbridge					TE	TE						TE	TE	TE					
WC 024.25	OWRD Gage 14076050						TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST	TE ST
WC 019.50	d/s Camp Polk bridge				TE		TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE
WC 018.25	d/s end of Camp Polk							TE		TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE
WC 009.25	u/s end of Rimrock Ranch															TE	TE	TE	TE	TE
WC 009.00	Rimrock Ranch												TE	TE	TE					
WC 008.75	d/s end of Rimrock Ranch															TE	TE	TE	TE	TE
WC 008.25	CRNG						TE													
WC 006.00	Road 6360	TE					TE ST	TE ST	TE ST			TE ST	TE ST	TE ST		TE ST	TE ST	TE ST	TE ST	TE ST
WC 003.00	u/s Alder Springs	TE																		
WC 001.50	d/s of Alder Springs		TE			TE	TE	TE				TE		TE	TE	TE	TE	TE	TE	TE
WC 001.00	Diamondback Meadow	TE												TE	TE	TE				
WC 000.25	Mouth of Whychus Creek		TE			TE		TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE

APPENDIX B Temperatures at given flows.

Whychus Creek at Sister's City Park (WC 024.25) at flows from 2.4 cfs to 221 cfs.

Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)
2	19.2	1.6	**57**	13.4	1.5	**112**	10.8	1.5	**167**	9.8	1.4
3	20.1	1.6	**58**	13.4	1.5	**113**	10.7	1.5	**168**	9.8	1.4
4	20.5	1.6	**59**	13.3	1.5	**114**	10.7	1.5	**169**	9.8	1.4
5	20.6	1.6	**60**	13.2	1.5	**115**	10.7	1.5	**170**	9.8	1.4
6	20.6	1.6	**61**	13.1	1.5	**116**	10.7	1.5	**171**	9.8	1.4
7	20.5	1.6	**62**	13.1	1.5	**117**	10.6	1.5	**172**	9.8	1.4
8	20.4	1.6	**63**	13.0	1.5	**118**	10.6	1.4	**173**	9.8	1.4
9	20.2	1.6	**64**	12.9	1.5	**119**	10.6	1.4	**174**	9.8	1.4
10	20.0	1.6	**65**	12.9	1.5	**120**	10.6	1.4	**175**	9.8	1.4
11	19.8	1.6	**66**	12.8	1.5	**121**	10.5	1.4	**176**	9.8	1.4
12	19.6	1.6	**67**	12.7	1.5	**122**	10.5	1.4	**177**	9.8	1.4
13	19.4	1.6	**68**	12.7	1.5	**123**	10.5	1.4	**178**	9.8	1.4
14	19.2	1.6	**69**	12.6	1.5	**124**	10.5	1.4	**179**	9.8	1.4
15	19.0	1.6	**70**	12.5	1.5	**125**	10.4	1.4	**180**	9.7	1.4
16	18.8	1.6	**71**	12.5	1.5	**126**	10.4	1.4	**181**	9.7	1.4
17	18.6	1.6	**72**	12.4	1.5	**127**	10.4	1.4	**182**	9.7	1.4
18	18.4	1.6	**73**	12.4	1.5	**128**	10.4	1.4	**183**	9.7	1.4
19	18.2	1.6	**74**	12.3	1.5	**129**	10.4	1.4	**184**	9.7	1.4
20	18.0	1.6	**75**	12.2	1.5	**130**	10.3	1.4	**185**	9.7	1.4
21	17.8	1.6	**76**	12.2	1.5	**131**	10.3	1.4	**186**	9.7	1.4
22	17.6	1.6	**77**	12.1	1.5	**132**	10.3	1.4	**187**	9.7	1.4
23	17.4	1.6	**78**	12.1	1.5	**133**	10.3	1.4	**188**	9.7	1.4
24	17.3	1.6	**79**	12.0	1.5	**134**	10.3	1.4	**189**	9.7	1.4
25	17.1	1.6	**80**	12.0	1.5	**135**	10.2	1.4	**190**	9.7	1.4
26	16.9	1.6	**81**	11.9	1.5	**136**	10.2	1.4	**191**	9.7	1.4
27	16.8	1.6	**82**	11.9	1.5	**137**	10.2	1.4	**192**	9.7	1.4
28	16.6	1.6	**83**	11.8	1.5	**138**	10.2	1.4	**193**	9.7	1.4
29	16.5	1.6	**84**	11.8	1.5	**139**	10.2	1.4	**194**	9.7	1.4
30	16.3	1.6	**85**	11.7	1.5	**140**	10.2	1.4	**195**	9.7	1.4
31	16.2	1.5	**86**	11.7	1.5	**141**	10.1	1.4	**196**	9.7	1.4
32	16.0	1.5	**87**	11.7	1.5	**142**	10.1	1.4	**197**	9.7	1.4
33	15.9	1.5	**88**	11.6	1.5	**143**	10.1	1.4	**198**	9.7	1.4
34	15.8	1.5	**89**	11.6	1.5	**144**	10.1	1.4	**199**	9.7	1.4
35	15.7	1.5	**90**	11.5	1.5	**145**	10.1	1.4	**200**	9.7	1.4
36	15.5	1.5	**91**	11.5	1.5	**146**	10.1	1.4	**201**	9.7	1.4
37	15.4	1.5	**92**	11.4	1.5	**147**	10.1	1.4	**202**	9.7	1.4
38	15.3	1.5	**93**	11.4	1.5	**148**	10.0	1.4	**203**	9.7	1.4
39	15.2	1.5	**94**	11.4	1.5	**149**	10.0	1.4	**204**	9.7	1.4
40	15.0	1.5	**95**	11.3	1.5	**150**	10.0	1.4	**205**	9.7	1.4
41	14.9	1.5	**96**	11.3	1.5	**151**	10.0	1.4	**206**	9.7	1.4
42	14.8	1.5	**97**	11.3	1.5	**152**	10.0	1.4	**207**	9.7	1.4
43	14.7	1.5	**98**	11.2	1.5	**153**	10.0	1.4	**208**	9.7	1.4
44	14.6	1.5	**99**	11.2	1.5	**154**	10.0	1.4	**209**	9.7	1.4
45	14.5	1.5	**100**	11.1	1.5	**155**	10.0	1.4	**210**	9.7	1.4
46	14.4	1.5	**101**	11.1	1.5	**156**	9.9	1.4	**211**	9.7	1.4
47	14.3	1.5	**102**	11.1	1.5	**157**	9.9	1.4	**212**	9.7	1.4
48	14.2	1.5	**103**	11.0	1.5	**158**	9.9	1.4	**213**	9.7	1.4
49	14.1	1.5	**104**	11.0	1.5	**159**	9.9	1.4	**214**	9.7	1.4
50	14.0	1.5	**105**	11.0	1.5	**160**	9.9	1.4	**215**	9.7	1.4
51	13.9	1.5	**106**	10.9	1.5	**161**	9.9	1.4	**216**	9.7	1.4
52	13.9	1.5	**107**	10.9	1.5	**162**	9.9	1.4	**217**	9.7	1.4
53	13.8	1.5	**108**	10.9	1.5	**163**	9.9	1.4	**218**	9.7	1.4
54	13.7	1.5	**109**	10.9	1.5	**164**	9.9	1.4	**219**	9.7	1.4
55	13.6	1.5	**110**	10.8	1.5	**165**	9.9	1.4	**220**	9.7	1.4
56	13.5	1.5	**111**	10.8	1.5	**166**	9.8	1.4	**221**	9.7	1.4

Whychus Creek at Road 6360 (WC 006.00) at flows from 2.4 cfs to 210 cfs (cubic equation).

Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)
2	22.1	1.8	**57**	18.5	1.8	**112**	16.1	1.7	**167**	15.1	1.7
3	23.7	1.9	**58**	18.4	1.8	**113**	16.1	1.7	**168**	15.0	1.7
4	24.4	1.9	**59**	18.3	1.8	**114**	16.1	1.7	**169**	15.0	1.7
5	24.6	1.9	**60**	18.3	1.8	**115**	16.0	1.7	**170**	15.0	1.7
6	24.6	1.9	**61**	18.2	1.8	**116**	16.0	1.7	**171**	15.0	1.7
7	24.6	1.9	**62**	18.2	1.8	**117**	16.0	1.7	**172**	15.0	1.7
8	24.5	1.9	**63**	18.1	1.8	**118**	16.0	1.7	**173**	15.0	1.7
9	24.3	1.9	**64**	18.0	1.8	**119**	15.9	1.7	**174**	15.0	1.7
10	24.1	1.9	**65**	18.0	1.8	**120**	15.9	1.7	**175**	14.9	1.7
11	24.0	1.9	**66**	17.9	1.8	**121**	15.9	1.7	**176**	14.9	1.7
12	23.8	1.9	**67**	17.9	1.8	**122**	15.9	1.7	**177**	14.9	1.7
13	23.6	1.9	**68**	17.8	1.8	**123**	15.8	1.7	**178**	14.9	1.7
14	23.4	1.9	**69**	17.8	1.8	**124**	15.8	1.7	**179**	14.9	1.7
15	23.2	1.9	**70**	17.7	1.8	**125**	15.8	1.7	**180**	14.9	1.7
16	23.0	1.9	**71**	17.7	1.8	**126**	15.8	1.7	**181**	14.9	1.7
17	22.9	1.8	**72**	17.6	1.8	**127**	15.7	1.7	**182**	14.9	1.7
18	22.7	1.8	**73**	17.6	1.8	**128**	15.7	1.7	**183**	14.9	1.7
19	22.5	1.8	**74**	17.5	1.8	**129**	15.7	1.7	**184**	14.8	1.7
20	22.3	1.8	**75**	17.5	1.8	**130**	15.7	1.7	**185**	14.8	1.7
21	22.2	1.8	**76**	17.4	1.8	**131**	15.7	1.7	**186**	14.8	1.7
22	22.0	1.8	**77**	17.4	1.8	**132**	15.6	1.7	**187**	14.8	1.7
23	21.9	1.8	**78**	17.3	1.8	**133**	15.6	1.7	**188**	14.8	1.7
24	21.7	1.8	**79**	17.3	1.8	**134**	15.6	1.7	**189**	14.8	1.7
25	21.6	1.8	**80**	17.2	1.7	**135**	15.6	1.7	**190**	14.8	1.7
26	21.4	1.8	**81**	17.2	1.7	**136**	15.6	1.7	**191**	14.8	1.7
27	21.3	1.8	**82**	17.1	1.7	**137**	15.5	1.7	**192**	14.8	1.7
28	21.2	1.8	**83**	17.1	1.7	**138**	15.5	1.7	**193**	14.8	1.7
29	21.0	1.8	**84**	17.1	1.7	**139**	15.5	1.7	**194**	14.7	1.7
30	20.9	1.8	**85**	17.0	1.7	**140**	15.5	1.7	**195**	14.7	1.7
31	20.8	1.8	**86**	17.0	1.7	**141**	15.5	1.7	**196**	14.7	1.7
32	20.7	1.8	**87**	16.9	1.7	**142**	15.4	1.7	**197**	14.7	1.7
33	20.6	1.8	**88**	16.9	1.7	**143**	15.4	1.7	**198**	14.7	1.7
34	20.5	1.8	**89**	16.9	1.7	**144**	15.4	1.7	**199**	14.7	1.7
35	20.3	1.8	**90**	16.8	1.7	**145**	15.4	1.7	**200**	14.7	1.7
36	20.2	1.8	**91**	16.8	1.7	**146**	15.4	1.7	**201**	14.7	1.7
37	20.1	1.8	**92**	16.7	1.7	**147**	15.4	1.7	**202**	14.7	1.7
38	20.0	1.8	**93**	16.7	1.7	**148**	15.3	1.7	**203**	14.7	1.7
39	19.9	1.8	**94**	16.7	1.7	**149**	15.3	1.7	**204**	14.7	1.7
40	19.8	1.8	**95**	16.6	1.7	**150**	15.3	1.7	**205**	14.6	1.7
41	19.7	1.8	**96**	16.6	1.7	**151**	15.3	1.7	**206**	14.6	1.7
42	19.6	1.8	**97**	16.6	1.7	**152**	15.3	1.7	**207**	14.6	1.7
43	19.6	1.8	**98**	16.5	1.7	**153**	15.3	1.7	**208**	14.6	1.7
44	19.5	1.8	**99**	16.5	1.7	**154**	15.2	1.7	**209**	14.6	1.7
45	19.4	1.8	**100**	16.5	1.7	**155**	15.2	1.7	**210**	14.6	1.7
46	19.3	1.8	**101**	16.4	1.7	**156**	15.2	1.7			
47	19.2	1.8	**102**	16.4	1.7	**157**	15.2	1.7			
48	19.1	1.8	**103**	16.4	1.7	**158**	15.2	1.7			
49	19.1	1.8	**104**	16.3	1.7	**159**	15.2	1.7			
50	19.0	1.8	**105**	16.3	1.7	**160**	15.1	1.7			
51	18.9	1.8	**106**	16.3	1.7	**161**	15.1	1.7			
52	18.8	1.8	**107**	16.3	1.7	**162**	15.1	1.7			
53	18.8	1.8	**108**	16.2	1.7	**163**	15.1	1.7			
54	18.7	1.8	**109**	16.2	1.7	**164**	15.1	1.7			
55	18.6	1.8	**110**	16.2	1.7	**165**	15.1	1.7			
56	18.5	1.8	**111**	16.1	1.7	**166**	15.1	1.7			

Whychus Creek at Road 6360 (WC 006.00) at flows from 2.4 cfs to 210 cfs (quartic equation).

Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)
2	23.1	1.8	**57**	17.9	1.7	**112**	15.1	1.6	**167**	14.2	1.6
3	23.7	1.8	**58**	17.8	1.7	**113**	15.0	1.6	**168**	14.2	1.6
4	24.1	1.8	**59**	17.8	1.7	**114**	15.0	1.6	**169**	14.2	1.6
5	24.3	1.8	**60**	17.7	1.7	**115**	15.0	1.6	**170**	14.2	1.6
6	24.4	1.8	**61**	17.6	1.7	**116**	15.0	1.6	**171**	14.2	1.6
7	24.4	1.8	**62**	17.5	1.7	**117**	14.9	1.6	**172**	14.1	1.6
8	24.3	1.8	**63**	17.5	1.7	**118**	14.9	1.6	**173**	14.1	1.6
9	24.2	1.8	**64**	17.4	1.7	**119**	14.9	1.6	**174**	14.1	1.6
10	24.1	1.8	**65**	17.3	1.7	**120**	14.8	1.6	**175**	14.1	1.6
11	24.0	1.8	**66**	17.2	1.7	**121**	14.8	1.6	**176**	14.1	1.6
12	23.8	1.8	**67**	17.2	1.7	**122**	14.8	1.6	**177**	14.1	1.6
13	23.7	1.8	**68**	17.1	1.7	**123**	14.8	1.6	**178**	14.1	1.6
14	23.5	1.8	**69**	17.0	1.7	**124**	14.8	1.6	**179**	14.1	1.6
15	23.4	1.8	**70**	17.0	1.7	**125**	14.7	1.6	**180**	14.1	1.6
16	23.2	1.8	**71**	16.9	1.7	**126**	14.7	1.6	**181**	14.1	1.6
17	23.0	1.8	**72**	16.8	1.7	**127**	14.7	1.6	**182**	14.1	1.6
18	22.9	1.8	**73**	16.8	1.7	**128**	14.7	1.6	**183**	14.1	1.6
19	22.7	1.8	**74**	16.7	1.7	**129**	14.6	1.6	**184**	14.1	1.6
20	22.5	1.8	**75**	16.6	1.7	**130**	14.6	1.6	**185**	14.1	1.6
21	22.3	1.8	**76**	16.6	1.7	**131**	14.6	1.6	**186**	14.1	1.6
22	22.2	1.8	**77**	16.5	1.7	**132**	14.6	1.6	**187**	14.1	1.6
23	22.0	1.8	**78**	16.5	1.7	**133**	14.6	1.6	**188**	14.1	1.6
24	21.9	1.8	**79**	16.4	1.7	**134**	14.6	1.6	**189**	14.1	1.6
25	21.7	1.8	**80**	16.4	1.7	**135**	14.5	1.6	**190**	14.1	1.6
26	21.5	1.7	**81**	16.3	1.7	**136**	14.5	1.6	**191**	14.1	1.6
27	21.4	1.7	**82**	16.3	1.7	**137**	14.5	1.6	**192**	14.1	1.6
28	21.2	1.7	**83**	16.2	1.7	**138**	14.5	1.6	**193**	14.1	1.6
29	21.1	1.7	**84**	16.2	1.7	**139**	14.5	1.6	**194**	14.1	1.6
30	20.9	1.7	**85**	16.1	1.7	**140**	14.5	1.6	**195**	14.1	1.6
31	20.8	1.7	**86**	16.1	1.7	**141**	14.4	1.6	**196**	14.1	1.6
32	20.7	1.7	**87**	16.0	1.7	**142**	14.4	1.6	**197**	14.1	1.6
33	20.5	1.7	**88**	16.0	1.7	**143**	14.4	1.6	**198**	14.1	1.6
34	20.4	1.7	**89**	15.9	1.7	**144**	14.4	1.6	**199**	14.1	1.6
35	20.3	1.7	**90**	15.9	1.7	**145**	14.4	1.6	**200**	14.1	1.6
36	20.1	1.7	**91**	15.8	1.7	**146**	14.4	1.6	**201**	14.1	1.6
37	20.0	1.7	**92**	15.8	1.7	**147**	14.4	1.6	**202**	14.1	1.6
38	19.9	1.7	**93**	15.7	1.7	**148**	14.3	1.6	**203**	14.1	1.6
39	19.8	1.7	**94**	15.7	1.7	**149**	14.3	1.6	**204**	14.1	1.6
40	19.6	1.7	**95**	15.7	1.7	**150**	14.3	1.6	**205**	14.1	1.6
41	19.5	1.7	**96**	15.6	1.6	**151**	14.3	1.6	**206**	14.1	1.6
42	19.4	1.7	**97**	15.6	1.6	**152**	14.3	1.6	**207**	14.1	1.6
43	19.3	1.7	**98**	15.5	1.6	**153**	14.3	1.6	**208**	14.1	1.6
44	19.2	1.7	**99**	15.5	1.6	**154**	14.3	1.6	**209**	14.1	1.6
45	19.1	1.7	**100**	15.5	1.6	**155**	14.3	1.6	**210**	14.2	1.6
46	19.0	1.7	**101**	15.4	1.6	**156**	14.3	1.6			
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48	18.8	1.7	**103**	15.3	1.6	**158**	14.2	1.6			
49	18.7	1.7	**104**	15.3	1.6	**159**	14.2	1.6			
50	18.6	1.7	**105**	15.3	1.6	**160**	14.2	1.6			
51	18.5	1.7	**106**	15.2	1.6	**161**	14.2	1.6			
52	18.4	1.7	**107**	15.2	1.6	**162**	14.2	1.6			
53	18.3	1.7	**108**	15.2	1.6	**163**	14.2	1.6			
54	18.2	1.7	**109**	15.2	1.6	**164**	14.2	1.6			
55	18.1	1.7	**110**	15.1	1.6	**165**	14.2	1.6			
56	18.0	1.7	**111**	15.1	1.6	**166**	14.2	1.6			

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 2011 UDWC retrofitted two barriers to provide fish passage, connecting two pairs of reaches and reducing total reaches below the natural upstream passage barrier to five. A third barrier was anecdotally determined not to be a barrier to fish passage. No new fish passage projects were completed in 2012. In 2013, dam removal and channel restoration was completed at a fourth former barrier to fish passage, and an agreement was reached to remove the last major barrier on Whychus Creek in 2014, leaving one upstream barrier to fish passage. With removal of the fourth barrier, an additional two reaches were connected, resulting in three reaches below the natural barrier. UDWC will continue to actively engage water rights holders to provide passage at the final two barriers by 2020. Removal of these barriers will provide access to an additional 14.5 river miles and restore connectivity along the entire length of stream habitat 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 down-stream fish passage by retrofitting or removing all fish passage barriers in Whychus Creek by 2020.

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 will indicate whether anadromous and resident fish are accessing that habitat. While physical barriers such as dams limit accessibility to fish habit, 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. 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 2013.

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 2012 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).

From 2010 to 2011 fish passage was restored at Barrier No. 3 at river mile 22.6, reducing the number of fragmented reaches to four and connecting two adjacent reaches to create a 1.6-mile reach. 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). In the absence of this barrier, 6.7 additional miles of habitat upstream of rm 15.5, and 22.2 miles total from the mouth of the creek to Barrier No. 2, were accessible to fish, and a total of four fragmented reaches remained below the natural barrier of Whychus Falls.

In 2011 an agreement was reached and a design completed to remove one additional barrier, Barrier No. 2 at rm 22.2, in 2012. A last-minute delay in federal funding forestalled implementation of this project during the 2012 construction season. The project was completed in October 2013, restoring fish passage at this point, increasing total miles of habitat accessible from the mouth of Whychus Creek to 23.8, and reducing fragmented reaches below natural barriers to three.

Over the course of 2013 UDWC and restoration partners continued to engage with water rights holders and landowners to plan fish passage restoration projects at remaining passage barriers. As of the end of 2013, an agreement was reached to address fish passage at a fifth barrier, No. 4, at rm 23.8, in 2014.

Table 1.
Passage barrier specifications and status as of 2013. 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.	
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

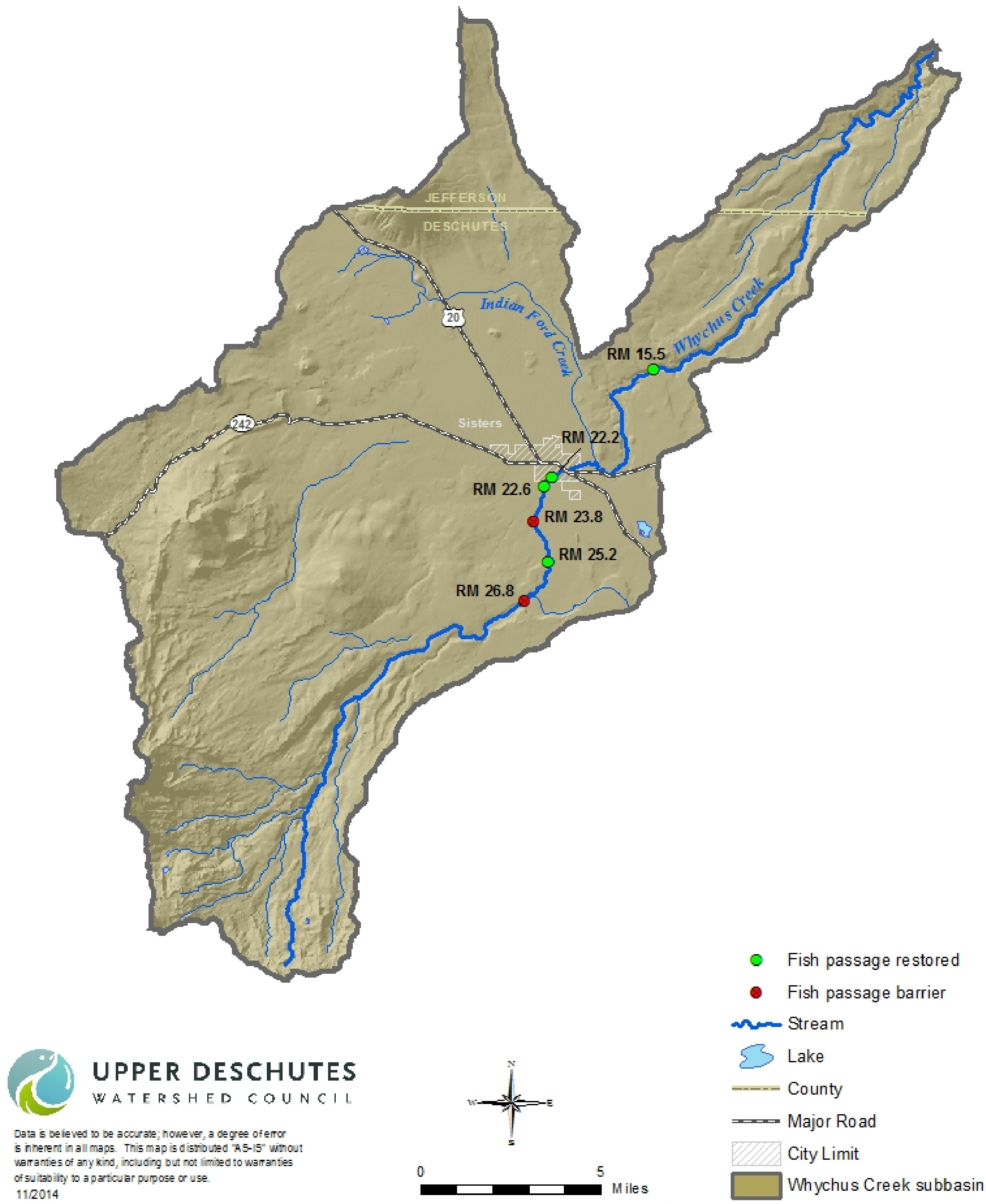

Figure 1.
In 2009, six barriers impaired stream connectivity on Whychus Creek between river miles 15.5 and 26.8. From 2009 to 2013 fish passage was restored or found to be intact at four of these barriers. UDWC and restoration partners aim to provide passage at each of these barriers by 2020.

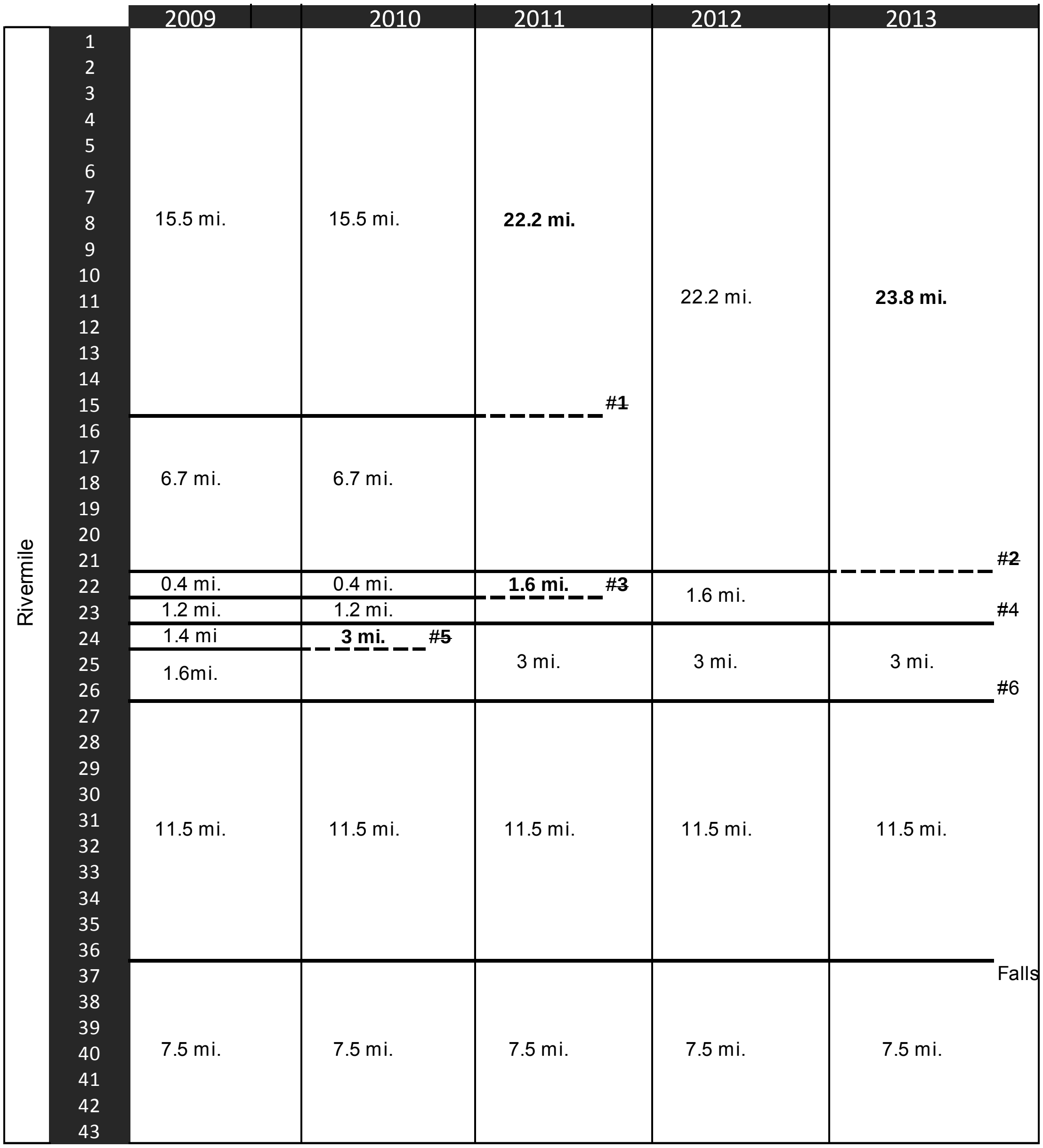

Figure 2.

Two barriers to fish passage remain as of 2013, reducing the number of fragmented reaches from a baseline of seven to three, varying in length from 3.0 to 23.8 miles. Whychus Creek Falls, located between river miles 36 and 37, is a 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 will continue to partner with water rights holders to develop and implement projects that will restore fish passage at the remaining two barriers by 2020. Removal of these barriers will provide access to an additional 14.5 river miles and restore connectivity along the entire length of stream habitat historically accessible to resident and anadromous species. Tracking miles of habitat newly opened to fish will continue to provide important context for the recovery of anadromous and resident fish populations in Whychus Creek.

References

Bergkamp G, McCartney M, Dugan P, McNeely J, Acreman M. 2000. Dams, Ecosystem Functions and Environmental Restoration Thematic Review II.1, prepared as an input to the World Commission on Dams. World Commission on Dams, Cape Town, South Africa.

Cote D, Kehler DG, Bourne C, Wiersma YF. 2009. A new measure of longitudinal connectivity for stream networks. Landscape Ecology 24:101–113.

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 Criteria. ODFW, Salem, Oregon. Unpublished.

ODFW (Oregon Department of Fish and Wildlife). 2009. Oregon Administrative Rules 635-412: Fish Passage. ODFW, Salem, Oregon.

OWRD (Oregon Water Resources Department). 2002. Whychus Creek Diversion Inventory. Oregon Water Resources Department, Bend, Oregon. Unpublished.

Wiens JA. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology 47: 501–515.

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, a tributary to Oregon's Deschutes River. 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 2009-2013 restoration partners screened four diversions, while water rights transactions reduced the total diversion rate by 27.24 cfs. In 2013, one diversion was inactive and a percentage of the water right was leased at a second diversion, further reducing the total diversion rate by 1.04 cfs. Together these actions reduced the cumulative unscreened diversion rate by 84% from 2009 to 2013, from 192.89 to 31.41 cfs. Potential for fish entrainment in irrigation diversions has thus been substantially reduced. UDWC and restoration partners remain committed to continuing to engage with water rights holders and landowners to eliminate all 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 2013.

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 2010 the total diversion rate was reduced to 178.02 cfs through streamflow restoration achieved by DRC that reduced flows diverted at TSID from 153.00 to 137.68. Two additional diversions, Edgington and Leithauser (Diversions No. 5 and 7), were screened to meet NMFS and ODFW criteria. Flows associated with these screens totaled 2.12 cfs, reducing the 2010 cumulative unscreened diversion rate to 175.45 cfs, still 98.5% of the total diversion rate. Construction of the fish screen at the Three Sisters Irrigation District diversion was completed in April 2011. At 137.68 cfs, the TSID diversion represents the single largest flow withdrawal on Whychus Creek. Completion of the TSID fish screen reduced the cumulative unscreened diversion rate on Whychus Creek to 37.77 cfs representing just 21% of the 2011 total diversion rate of 178.02 cfs. A detailed phone conversation with the water rights holder for Diversion No. 11 established the diversion, a small-diameter pipe, likely posed a minimal risk for fish entrainment (R. Houston, personal communication 2012); the presence of a screen, although unknown to meet screening criteria, was confirmed by the OWRD Basin Watermaster (J. Giffin, personal communication, July 2014).

Following a 2011 agreement with water rights holders, in 2012 DRC completed water rights transactions that reduced the diversion rate at the Lazy Z/Uncle John diversion from 9.3 cfs (higher than reported in the 2009 baseline inventory) to 0.00 and transferred 1.61 cfs instream; ongoing water rights transactions will transfer an additional 7.69 cfs of Lazy Z water rights to TSID. The Lazy Z/Uncle John diversion (Diversion No. 3, rm 25.5) was decommissioned and the point of diversion consolidated with TSID, eliminating the risk of fish entrainment at this site. This consolidation reduced the number of diversions not meeting screening criteria to eight, and reduced unscreened flows by 9.3 cfs. Between water rights transactions and screening of irrigation diversions, the cumulative unscreened diversion rate was reduced to a new low of 32.25 in 2012. Water rights transactions further reduced diversion rates by an additional 1.00 cfs at Leithauser, however, because Leithauser had already been screened, this reduction did not affect the cumulative unscreened diversion rate.

Diversion No. 1, Plainview, was abandoned and the canal for Diversion No. 2, McCallister, breached to allow access to irrigation flows. In 2012 the USFS filled the abandoned diversion to construct a trail, incidentally eliminating risk of fish entrainment at the former diversion.

Although no fish screening projects were implemented in 2013, the holder of the water rights associated with diversion No. 12 did not divert, and the holder of the water rights associated with diversion No. 13 leased 0.24 cfs instream, reducing the cumulative flow diverted through unscreened diversions in 2013 by 1.04 cfs. During 2013 UDWC engaged in planning efforts for two large-scale restoration projects that will ultimately screen or remove a total of three diversions in Whychus Creek and reduce the cumulative rate of diversion through unscreened diversions by 6.28 cfs. The Whychus Canyon Restoration Project will screen or remove diversions No. 12 and No. 13, in project reaches 2 and 4, respectively, to eliminate the risk of fish entrainment at these sites. The Whychus Canyon Restoration Plan was completed and peer reviewed in 2013, with implementation in reach 4 scheduled for 2016. The Whychus Floodplain project will eliminate the historic diversion at Pine Meadow Ranch and replace it with a pump outfitted with a fish screen to, among other objectives, accomplish screening at Diversion No. 6. The Whychus Floodplain Restoration Plan was completed and peer reviewed in 2013, and installation of the new pump and screen are scheduled for April 2014.

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 2013 conversations with water managers and water rights holders to address fish screening at Diversion No. 2 were ongoing. Screening or decommissioning the four diversions currently in various stages of planning will reduce the total number of unscreened diversions to three.

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 and one diversion was decommisioned between 2009 and 2013, leaving seven diversions unscreened.

2002-2009 Baseline data											
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	2013 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	133.68	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	5.00		No	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		No	Meyer. Fish screening assessed by OWRD
No. 12	9/24/2002	11.20	Gravity	0.68	No	N/A	No	0.68		No	Remund. Not diverted in 2013; 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 2013; diversion will be removed or screened within Whychus Canyon project
			Baseline Diversion Total	**193.34**			**2013 Diversion Total**	**166.54**			
			Baseline Unscreened Total	**192.89**			**2013 Unscreened Total**	**31.41**			

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

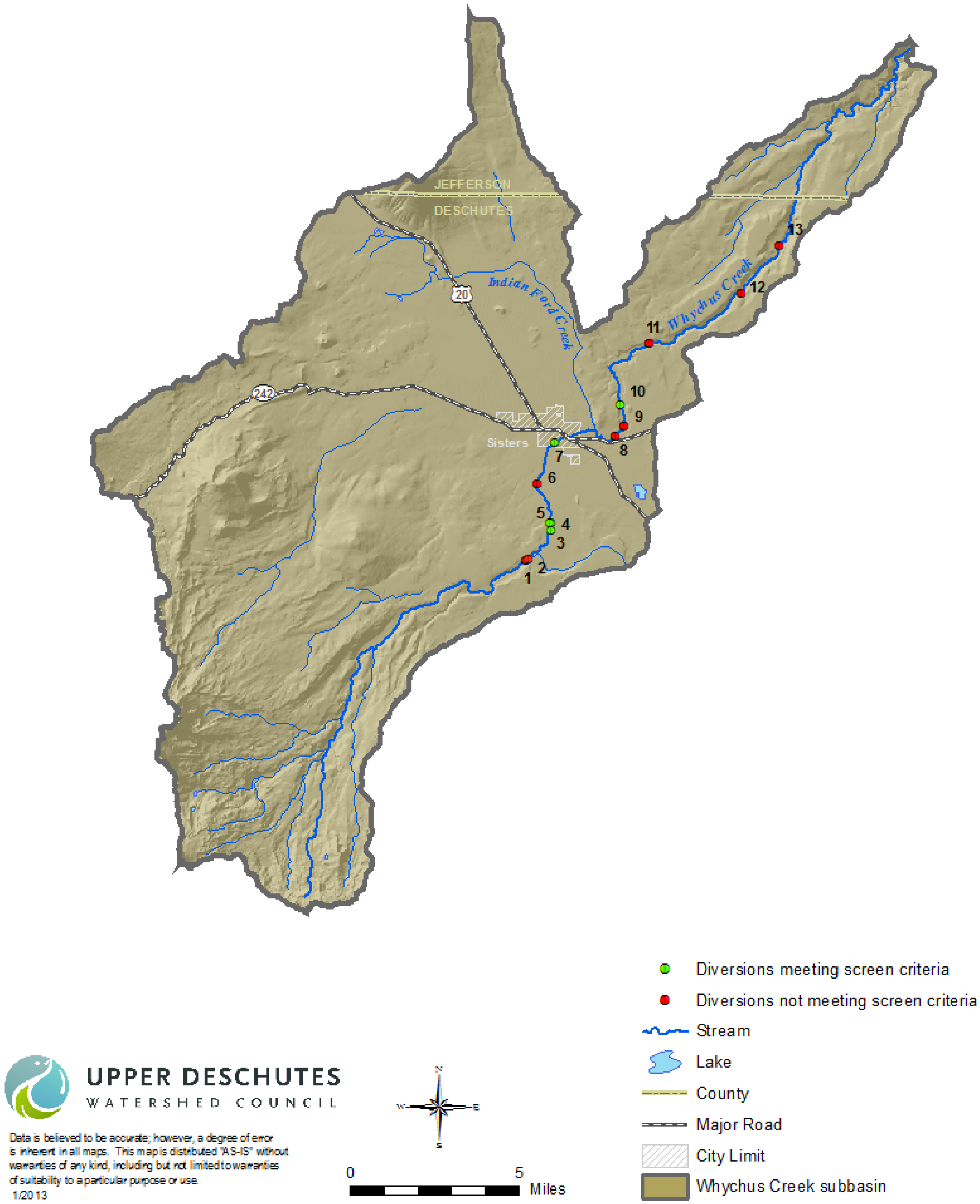

Figure 1.
In 2009 UDWC identified 13 irrigation diversions on Whychus Creek of which 12 did not meet federal and state criteria for fish screens (NMFS 2008). From 2009 to 2013 cumulative flows diverted through unscreened diversions were reduced by 84%, to 31.41 cfs. Although the screen at Diversion no. 11 has not been verified to meet screening criteria and is therefore shown as such, 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 of characterizing fish entrainment potential risks 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 are not seeking 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.

As of the close of 2013, the cumulative unscreened diversion rate had been reduced by 84%, from 192.89 in 2009 to 31.41 cfs in 2013. Potential for fish entrainment in irrigation diversions has thus been substantially reduced, owed in large part to the progressive practices of TSID management. UDWC and restoration partners remain committed to continuing to engage with water rights holders and landowners to eliminate all risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

References

Carlson AJ, Rahel FJ. 2007. A basinwide perspective on entrainment of fish in irrigation canals. Transactions of the American Fisheries Society 2007; 136: 1335-1343 doi: 10.1577/T06-111.1

Gale SB, AV Zale, and CG Clancy. 2008. Effectiveness of fish screens to prevent entrainment of westslope cutthroat trout into irrigation canals. North American Journal of Fisheries Management 28: 1541-1553.

Houston R. 2012. Personal communication. Executive Director, Upper Deschutes Watershed Council, Bend, Oregon.

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

OWRD (Oregon Water Resources Department). 2002. Whychus Creek Senior Water Rights by Diversion Inventory.

Riehle M. 2009. Personal communication. Fish Biologist, US Forest Service, Sisters Ranger District, Sisters, Oregon.

Roberts JJ, Rahel FJ. 2008. Irrigation canals as sink habitat for trout and other fishes in a Wyoming drainage. Transactions of the American Fisheries Society.137:951-961.

Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, and 2011-2013

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Abstract

Aquatic macroinvertebrates were sampled in 2005, 2009, 2011, 2012, and 2013 at 10-13 sites in Whychus Creek (RM 30.25 to RM 0.5) to determine biological conditions and assess the effects of restoration on stream biota. The benthic macroinvertebrate community in Whychus Creek showed the greatest magnitude of change from 2005 to 2009, after which there was increasing stabilization. PREDATOR model and multi-metric Index of Biological Integrity (IBI) scores indicated parallel trends in biotic condition in downstream reaches but presented differing portrayals of conditions in mid-reach and upstream sites. PREDATOR scores for downstream reaches indicated significant improvement in biotic conditions from 2005 to 2009 followed by little change through 2013. IBI scores for downstream reaches increased significantly from 2005 to 2011 with a corresponding improvement from moderately to slightly impaired condition, and maintained the improved biological condition through 2013 despite lower mean scores in 2012 and 2013. PREDATOR scores for mid-reach sites fluctuated, with significantly lower scores indicating poor conditions in 2011-2013 compared to good conditions in 2005 and 2009. IBI scores for mid-reach sites tracked PREDATOR scores, but suggested biological conditions remaining slightly impaired in contrast to the marked decrease in conditions indicated by the PREDATOR model. PREDATOR and IBI scores at mid-reach sites are likely to continue to change as the stream responds to the channel relocation at Camp Polk in 2012. PREDATOR scores indicating poor biotic conditions in upstream sites did not differ significantly among years, while mean IBI scores for upstream sites increased significantly from 2005 to 2012, indicating an improvement in biotic condition from slightly to minimally impaired, but fell in 2013 to suggest a return to a slightly impaired biological condition. Increased mean IBI scores among upstream reaches may have been driven by an increase in richness, relative diversity, and abundance of sensitive EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) at upstream sites from 2005 to 2012. Mean optima values indicating tolerance for temperature and fine sediments were consistently highest for downstream sites and decreased through mid-reach and upstream sites. Although mean temperature optima for replacement v. missing taxa were significantly different in only a single year, mean temperature optima for the entire macroinvertebrate assemblage decreased steadily and significantly in all reaches from 2005 to 2011, and fell again from 2012 to 2013 at the downstream and mid-reach sites expected to be most affected by streamflow restoration. The latter trend was also observed for fine sediment optima although was less pronounced. Fine sediment optima of replacement taxa were significantly lower than optima of missing taxa in all years. Although this result could indicate the stream is sediment-deprived, it also calls into question the applicability of the PREDATOR model observed and expected taxa assemblage to Whychus Creek. Five years of data suggest that following an initial shift in composition the macroinvertebrate community may be stabilizing in an assemblage that reflects slightly to moderately impaired or fair to poor conditions. The persistence of a macroinvertebrate assemblage indicating degraded biological conditions may be an artifact of historic

habitat alterations that have yet to be remediated by restoration, or it may suggest that additional stressors continue to influence biological conditions and the resulting macroinvertebrate communities.

Project Background

Biomonitoring in Whychus Creek

Biomonitoring evaluates the biological health of a water body by examining its biotic communities, including plants, amphibians, algae, diatoms, fish, or invertebrates (Rosenberg & Resh 1993, Karr & Chu 1999). Habitat degradation and anthropogenic stressors alter the structure of these communities, according to individual species' sensitivity or tolerance to different stressors and their ability to persist under different conditions. Benthic macroinvertebrates are ideal subjects for biomonitoring because:

- They are important in nutrient and energy cycling and are a critical part of the food web. For example, restoration done to help recovery of native fish populations is unlikely to be successful in the absence of an aquatic invertebrate food base, as the quantity and quality of prey items affects juvenile growth and survival (Gibson 1993).
- The life history and relatively limited mobility of many groups confines them to water for most or all of their life cycle; if conditions become unsuitable, they will die and/or migrate out of the area.
- They exhibit a range of responses to human-induced stressors, and changes in different groups can reflect the effects of temperature, substrate, habitat complexity, or flow.
- Their short generation time allows changes in community structure to be detected rapidly following a disturbance.
- They are ubiquitous, abundant, and unlikely to be completely absent from any but the most egregiously polluted water bodies.
- Sampling and identification are relatively straightforward, standardized, and cost-effective.

Biomonitoring may be done to determine the baseline biological conditions of selected communities, investigate the impacts of a disturbance or pollutant, or assess changes following restoration projects. 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, such as expected changes in macroinvertebrate community composition as a stream flows from high-elevation headwaters to valleys (i.e. Vannote *et al*. 1980), or as high spring flows generated by snowmelt and rain decrease to the groundwater-sustained low flows of late summer and autumn. Benthic macroinvertebrates are key biological indicators, as community composition at sampling sites from near the headwaters to the mouth can change over time in response to reach- and catchment-scale land management practices and habitat restoration activities (Albertson *et al*. 2011).

Whychus Creek experienced significant habitat degradation in past decades from surrounding land use practices, including dewatering for irrigation, channelization, grazing, and stream-side development. This project is part of a long-term monitoring effort to evaluate changes in watershed conditions in Whychus Creek as both large scale and site-specific restoration projects are implemented (Upper Deschutes Watershed Council 2009). Xerces Society worked with Upper Deschutes Watershed Council to collect benthic macroinvertebrate samples along Whychus Creek in 2005, 2009, 2011, 2012, and 2013 from sites spanning RM 30.25 to RM 0.5. Sampling in 2005 was done prior to large-scale habitat restoration and before some stream flow restoration. This provided baseline data on existing macroinvertebrate communities along the stream, which in previous years would frequently go dry in summer. Sampling was repeated in 2009 and 2011 to assess the macroinvertebrate community after large scale stream flow restoration had been accomplished, including the restoration of about 20 cfs of water to the creek flow, but prior to the planned restoration of channelized reaches at Camp Polk Meadow. Sampling in 2012 and 2013 occurred after the stream at Camp Polk Meadow was re-meandered; monitoring and maintenance of this restoration site is ongoing.

Biotic assessment techniques

Predictive models

Biological community assessment is often done using two major analytical approaches: predictive models and multimetric Indices of Biological Integrity (IBI). Predictive models compare the macroinvertebrate community at a given site to the community at reference streams or streams in best-available condition with similar physical, chemical, and biological characteristics in the same region (Wright *et al*. 2000). The predictive model PREDATOR (Predictive Assessment Tool for Oregon; Hubler 2008) was developed for two major regions in Oregon: the Marine Western Coastal Forest (Willamette Valley and Coast Range ecoregions) and the Western Cordillera and Columbia Plateau (Klamath Mountain, Cascades, East Cascades, Blue Mountains, and Columbia Plateau ecoregions). The model calculates the ratio of taxa observed at a sampling site to taxa expected at that site if no impairment existed (O over E), based on community data collected previously at a large number of reference streams. In general, an O/E value of less than one indicates loss of common taxa, while values greater than one indicate taxa enrichment, potentially in response to pollution or nutrient loading. The model output also generates O/E scores for individual taxa at each sampling site, allowing specific taxa loss and replacement to be investigated.

While predictive models are often considered to be more sensitive and accurate than multimetric assessments (see below), it should be noted that the PREDATOR model has not been re-calibrated since it was created using streams 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 WC+CP (Western Cordillera+Columbia Plateau) predictive model applicable to the Cascades is known to have lower precision than the MWCF (Marine Western Coastal Forest) model developed for the Willamette Valley and Coast Range (Hubler 2008; Hubler, pers. comm. 2013). In addition, while a PREDATOR score indicating "poor" can be reliably assumed to reflect biologically disturbed conditions, a site that scores as "fair" is less certain to have an O/E score truly outside of the reference distribution, and repeated sampling at the site along with additional habitat and water quality assessments are recommended (Hubler 2008).

Biological Indices

Biological indices rate a combination of community attributes (metrics) that respond predictably to human-induced stressors (Karr & Chu 1999). Individual metrics are scored and summed to generate a total index of biological integrity (IBI) value that reflects the biological condition of a site. Multimetric biological indices have been developed in Oregon for use with macroinvertebrate stream taxa identified either to family (Level 2 assessment) or to genus and species (Level 3 assessment; OWEB 2003). Genus- and species-level identification is preferred over the broader family-level taxonomy for IBI assessment, as a single family often contains individual genera that differ in tolerances and response to disturbances.

Metrics are based on the rationale that a less disturbed, healthier stream system has greater biodiversity and thus will be higher in both overall taxa diversity and in diversity of sensitive taxa such as mayflies, caddisflies, and stoneflies (Norris & Georges 1993, Barbour *et al*. 1996). However, diversity metrics must be treated with caution; moderate levels of disturbance may actually result in an increase in diversity before the disturbance becomes severe enough for the community to be wholly degraded (intermediate disturbance hypothesis; Connell 1978, Ward & Stanford 1983, Townsend *et al*. 1997), as varying stressors cause continuous local extirpation of taxa followed by re-invasion of depleted niches. A healthy system is also expected to have a more balanced composition of taxa, such that a few genera or species do not dominate. A large abundance of a small number of taxa is indicative of impaired conditions and environmental stressors, as the macroinvertebrate community becomes dominated by one or a few tolerant groups (Plafkin *et al*. 1989, Barbour *et al*. 1996).

Unlike the regionally-targeted PREDATOR models, the macroinvertebrate IBI currently in use in Oregon was developed from a smaller dataset and does not consider regional differences to the same extent as the predictive models (Hubler 2008 and pers. comm.). However, regular assessment of stream condition using the macroinvertebrate IBI enables detection of changes and trends in macroinvertebrate community composition, especially among sensitive taxa.

Methods

Sampling Sites

Ten sites in Whychus Creek were sampled in 2005 and 2009, 13 sites were sampled in 2011, 11 sites were sampled in 2012, and 12 sites were sampled in 2013 (Table 1). Duplicate samples were taken at one to two sites each year for quality assurance. The sites selected are historic water quality monitoring stations where physical, chemical, and/or biological data has been collected previously, and where temperature dataloggers have been placed. Of the 12 sites sampled in 2013, five were sampled across all five years of the study, four were sampled across 2009-2013, two were sampled across 2011-2013, and one was a new site added in 2013. During the course of the study some sites were changed or replaced (generally with sites at a nearby location) due to access, changes in land use, or decisions by watershed council staff that they did not provide information that was critical or relevant at the watershed scale. The reach at RM 30.25 was not sampled after 2011 because the creek in that region was too deep, wide, and fast for standard wadeable stream sampling techniques to be appropriate, and kicknet samples taken there were not truly comparable to samples from other sites. Overall, sampling sites are distributed broadly along Whychus Creek into downstream (RM 0.5- 9.5), mid-stream (RM 18-19.5), and upstream reaches (RM 23.5 - 30.25). A map of the sampling sites is shown in Appendix A.

Volunteer Training & Sampling Techniques

Training

Volunteer training and stream sampling was done in the same index period across all years, ranging from August 17-20, depending upon when the Saturday fell. Watershed council staff and volunteers assembled at City Park (Sisters, OR) and were trained by Xerces staff in standardized macroinvertebrate monitoring protocols for Oregon's wadeable streams (OWEB 2003). The sampling protocol was demonstrated and each item on the data sheet was explained (see Appendix B for data sheet). A handout with detailed step-by-step sampling instructions was provided, as well as field guides to Northwest stream macroinvertebrates (Adams *et al*. 2003) and freshwater mussels (Nedeau *et al*. 2009), although volunteers were not expected to identify any organisms. The group was then divided into teams of three to four people, each of which received the following equipment: D-frame kick net with 500 μm mesh, metal 500 μm sieve, a square of 500 μm mesh Nitex membrane to line the sieve (in 2012

& 2013), plastic squirt bottle, forceps, thermometer, fiberglass tape measure, 10-gallon plastic bucket, hand lens, 1-liter Nalgene sample jars containing 80% ethanol, datasheets, jar labels, and clipboard.

Table 1. Whychus Creek sampling sites, 2005-2013

Site ID	Description	Coordinates	Year sampled
WC00050	RM 0.50	44.45682, -121.34028	2005
WC00150	RM 1.5, d/s Alder Springs	44.44491, -121.34543	2009, 2011-2013
WC00300[a]	RM 3, u/s Alder Springs	44.43458, -121.35976	2005
WC00600[b,d]	RM 6, u/s Rd 6360	44.40412, -121.40259	2005, 2009, 2011-2013
WC00650	RM 6.5, u/s Rd 6360 crossing	44.396799, -121. 404268	2013
WC00875[d]	RM 8.75, Rimrock Ranch d/s	44.391278, -121.406182	2011-2013
WC00900	RM 9, Rimrock Ranch	44.384198, -121.407892	2005, 2009, 2011-2013
WC00950[e]	RM 9.5, Rimrock Ranch u/s	44.371534, -121.415865	2011-2013
WC01800	RM 18	44.328342, -121.494534	2005
WC01825	RM 18.25, d/s end DBLT property	44.32781, -121.495406	2009, 2011-2013
WC01850	RM 18.5, DBLT property	44.326601, -121.500229	2009, 2011-2013
WC01900	RM 19, DBLT property	44.321523, -121.507461	2005, 2009, 2011-2013
WC01950	RM 19.5, d/s Camp Polk Bridge on DBLT property	44.318741, -121.514961	2009, 2011-2013
WC02350	RM 23.5, Perit Huntington Rd.	44.29066, -121.53064	2005
WC02425[c,e]	RM 24.25, City Park, d/s gauge	44.287806, -121.544229	2005, 2009, 2011-2013
WC02600[c]	RM 26, 4606 Rd. footbridge	44.2730592, -121.555297	2005, 2009, 2011-2013
WC02650	RM 26.5, d/s TSID	44.256434, -121.550692	2011
WC02700	RM 27, u/s TSID	44.250744, -121.549892	2011
WC03025	RM 30.25, OWRD gauge	44.233647, -121.567105	2005, 2009, 2011

Superscripted letters indicate sites where duplicate samples were taken in each year for quality control purposes; [a] 2005 duplicate; [b] 2009 duplicate; [c] 2011 duplicate; [d] 2012 duplicate; [e] 2013 duplicate

Stream sampling & sample processing

Macroinvertebrate samples were collected from riffle habitats at each site according to standardized protocols (OWEB 2003). Sampling reaches were calculated as 40 times the average wetted width of the stream at the desired sampling point. In 2005 and 2009, volunteers calculated the wetted width and paced out the sampling reach length; in 2011-2013, watershed council staff calculated wetted widths and flagged the upstream and downstream extent of each reach for volunteers prior to sampling.

Eight randomly selected riffles within each stream reach were sampled. Each sample consisted of eight net sets, each collected from a one-foot by one-foot area using a 500 μm D-frame kick net. In reaches with fewer than eight riffles, two kicknet samples were taken in each of four riffles in the reach. Large rocks and debris in the sampling area were first rinsed into the net to dislodge and collect any clinging organisms and set aside. The substrate was then disturbed thoroughly using a boot heel or hand to a depth of ~10 cm for 30-60 seconds. The eight individual net samples at each site were pooled into one bucket; large debris was rinsed and removed, and any vertebrates such as fish were carefully replaced in the stream. Sample material was concentrated by being poured through a 500 μm sieve and the composited material was placed into 1-liter Nalgene jars with 80% ethanol as a preservative. In 2012 and 2013, sieves were lined with flexible 500 μm Nitex membrane, which can be lifted from the sieve and the collated sample material rinsed more easily and completely into the sample jar.

If the sample contained excessive amounts of sand and gravel, it was elutriated by adding water to the sample bucket, swirling it to allow lighter organic material including macroinvertebrates to be suspended, then pouring the suspended material on the sieve. After two to three such rinses, the organic material was placed in sample jars separate from the mineral material, to prevent the organisms from being damaged during transport, but all sample material from each site was retained and subsequently examined. Jars were filled no more than halfway with sample material to ensure preservation, and the ethanol was replaced within 48 hours to maintain 80% concentration, as water leaches from the sample material 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 (Appendix B).

Macroinvertebrate Identification

Samples collected in 2005 were identified by Aquatic Biology Associates, Inc. (ABA; Corvallis, OR); those collected in 2009, 2011, and 2012 were identified by ABR, Inc. Environmental Research & Services (Forest Grove, OR); and 2013 samples were identified by Cole Ecological, Inc. (a company formed in 2013 by Mike Cole, the taxonomist who worked with us in previous years at ABR, 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.

Overall, organismal abundance has been high enough that the target count was achieved for most samples; those with lower abundance are generally more upstream sites, which is not unusual for small cold headwater streams (Crunkilton & Duchrow 1991, Lillie *et al*. 2003). In 2005, only one site had fewer than 500 organisms (RM 30.25; 397 organisms total). In 2009, the 500 organism target of was attained for all sampling sites, but in 2011, the target count was attained for only 7 of the 13 sites sampled, with upstream sites again having lower abundance. In 2012, the target count was attained for eight of the 11 samples, with only one of the more upstream sites having a slightly lower abundance (RM 18.25 sample, 475 organisms). In 2013, the target count was attained for 12 of the 14 samples taken, with only the sample from RM 26 and a duplicate sample taken at RM 24.25 yielding fewer than 500 organisms.

All organisms picked from the samples were identified to genus and species, where possible. When a specimen was too immature for key taxonomic characters to be fully developed or visible, identification was done only to the family level.

Data Analysis

The benthic macroinvertebrate community was assessed using multimetric and multivariate techniques. Sampling data for each year was analyzed using the PREDATOR predictive model for the Western Cordillera + Columbia Plateau (WC+CP; Hubler 2008). Observed over expected (O/E) scores associated with a probability of capture (P_c) > 0.5 were used (i.e. the model uses only invertebrates with over 50% likelihood of being collected at reference sites). The O/E benchmarks for describing biological conditions in the WC+CP model are:

Poor (most disturbed): O/E = $\leq$0.78

Fair (moderately disturbed): O/E = 0.79 – 0.92

Good (least disturbed): O/E = 0.93 – 1.23

Enriched: O/E = >1.23

PREDATOR scores are generated based on a site habitat file and a sample data file which are loaded into the model software at the Western Center for Monitoring & Assessment of Freshwater Ecosystems (http://cnr.usu.edu/wmc/htm/predictive-models/predictivemodelsoftware). Model output includes a site test result, which indicates whether the habitat data falls within the model parameters (i.e. is the appropriate model being used for the site); an O/E score for each sampling site, which indicates biological condition; a probability matrix showing taxa that are expected to occur at each site but are absent (missing taxa) and observed taxa that are not expected to occur at the site (replacement taxa); and a taxon occurrence summary that shows the mean probability of capture of each taxon, the total number of sites at which the taxon is expected, and the number of sites at which it was collected.

A DEQ dataset containing optima values determined for seasonal maximum temperatures and percent fine sediments for macroinvertebrate taxa (Huff *et al*. 2006) was used to investigate whether differences in temperature or sediment conditions could explain missing or replacement taxa among sampling sites. These optima were also used to calculate weighted average values for temperature and sediment optima for the macroinvertebrate assemblage at each site in each sampling year.

Biological condition was also assessed using the Oregon Level 3 (genus and species level) multimetric Index of Biological Integrity (IBI; OWEB 2003), which consists of 10 metrics. The raw value of each metric is calculated and accorded a corresponding scaled score of 5, 3, or 1, with higher scores indicating better biological condition (i.e. closer to reference values). Scaled values for individual metrics are summed to yield a single site-specific IBI score, which can reflect a biological condition of minimal (>39), slight (30-39), moderate (20-29), or severe impairment (<20). Individual metrics were calculated for each sampling site and used to determine the IBI score and condition. Metrics are shown below; the first number shows the possible data range, and the number in parentheses is the scaled IBI score that corresponds to the raw value:

- Taxa richness (# of taxa at site): >35 (5), 19-35 (3), <19 (1)
- Ephemeroptera (mayfly) richness: >8 (5), 4-8 (3), <4 (1)
- Plecoptera (stonefly) richness: >5 (5), 3-5 (3), <3 (1)
- Trichoptera (caddisfly) richness: >4 (5), 2-4 (3), <2 (1)
- Number of sensitive taxa: >4 (5), 2-4 (3), <2 (1)
- Number of sediment-sensitive taxa: >2 (5), 1 (3), 0 (1)
- % dominance of the top taxon: <20 (5), 20-40 (3), >40 (1)
- % tolerant taxa: <15 (5), 15-45 (3), >45 (1)
- % sediment-tolerant taxa: <10 (5), 10-25 (3), >25 (1)
- Modified Hilsenhoff Biotic Index (MHBI; a measure of pollution tolerance): <4.0 (5), 4-5 (3), >5.0 (1)

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 was conducted on a Bray-Curtis similarity matrix of square-root transformed abundance data to investigate macroinvertebrate community similarity between sites and across years. 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.

Results and Discussion

PREDATOR analysis

Site test results

The site test results in each year indicated that all predictor variables for the test samples were within the experience of the WC+CB model, and thus the PREDATOR results can be considered valid.

Site O/E

PREDATOR observed/expected scores (O/E) reflected a general improvement in biological condition from 2005 to 2009, after which conditions appeared to either stabilize or decrease slightly (Table 2 & Figure 1). O/E scores at sites sampled in 2005 rated two sites as good, with one of these sites just slightly above the transition from good to enriched, four sites as poor, and four as fair. In 2009, PREDATOR analysis ranked three sites as good and five as fair, with only the two most upstream sites scoring as poor (at RM 26 and RM 30.25). In 2011, four sites scored as fair and the remaining 10 sites scored as poor. In 2012, five of the sampling sites received O/E scores indicating fair conditions, and the remaining six scored as poor condition. The situation was similar in 2013, with five of the 12 sampling sites scoring as fair, and the remaining scoring as poor.

Table 2. PREDATOR O/E scores. Colored cells show biological conditions that are enriched (orange), good (least disturbance; green), fair (moderate disturbance; blue), or poor (most disturbed; red).

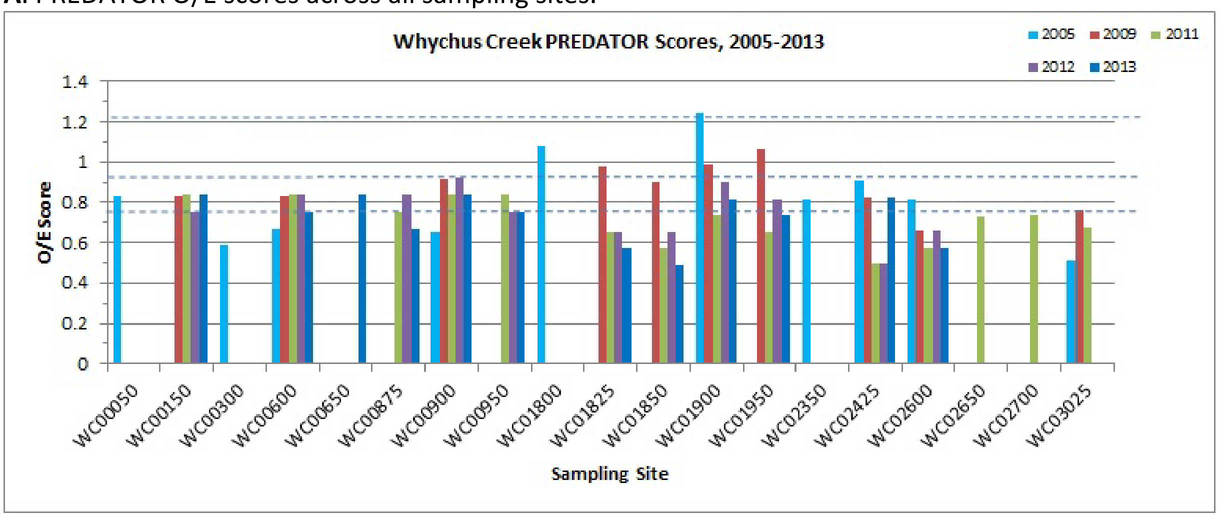
Site	2005	2009	2011	2012	2013
WC00050	0.832696	N/A	N/A	N/A	N/A
WC00150	N/A	0.832498	0.836044	0.75244	0.836044
WC00300	0.585774	N/A	N/A	N/A	N/A
WC00600	0.668248	0.831966	0.838606	0.838606	0.754745
WC00650	N/A	N/A	N/A	N/A	0.838486
WC00875	N/A	N/A	0.753175	0.836861	0.669489
WC00900	0.648243	0.91514	0.836575	0.920232	0.836575
WC00950	N/A	N/A	0.840275	0.756247	0.752917
WC01800	1.079496	N/A	N/A	N/A	N/A
WC01825	N/A	0.981688	0.6528	0.6528	0.5712
WC01850	N/A	0.89988	0.569871	0.651282	0.488461
WC01900	1.239275	0.981938	0.7344	0.897601	0.816
WC01950	N/A	1.0638	0.652978	0.816223	0.734601
WC02350	0.815577	N/A	N/A	N/A	N/A
WC02425	0.906197	0.820987	0.49403	0.49403	0.823383
WC02600	0.815082	0.65668	0.576268	0.658592	0.576268
WC02650	N/A	N/A	0.730367	N/A	N/A
WC02700	N/A	N/A	0.737997	N/A	N/A
WC03025	0.508901	0.76315	0.676991	N/A	N/A

Results varied among sites sampled for multiple years. For example, the O/E scores at RM 9 indicated poor conditions in 2005 but fair in each of the following years, though the value fluctuated from year to year, while at RM 18 and 18.25, initial fair to good O/E scores in 2009 decreased in 2011-2013 to scores indicating poor conditions. Because individual site scores varied to different degrees across the years, we examined PREDATOR scores in all sampling years based on the overall stream reach in which the sampling site is located.

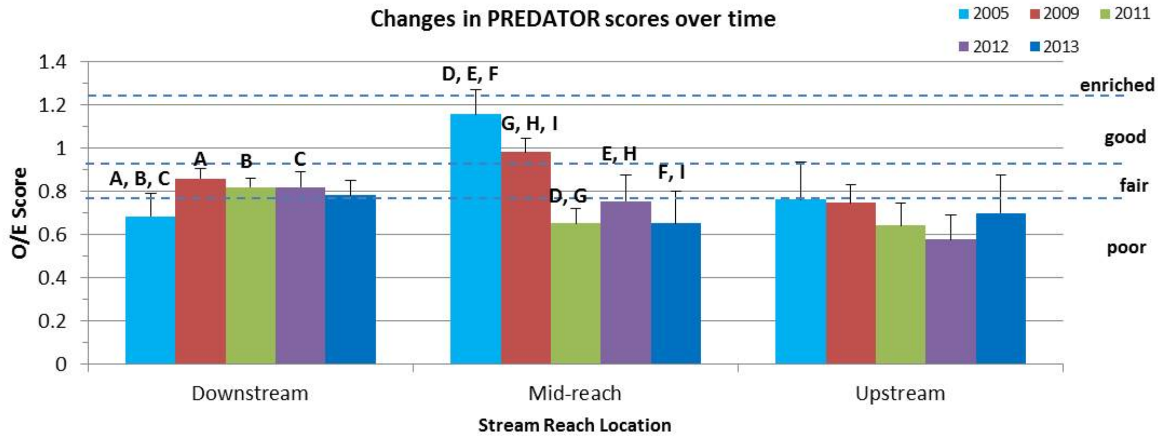
The amount of variation in O/E scores differs in different stream reaches (Figure 1). Sampling sites located in the downstream reaches of Whychus Creek (RM 0.5 to 9.5) show the greatest overall stability in O/E scores and biological condition. The mean PREDATOR score for downstream reaches in 2005 indicated overall poor conditions; however, the mean PREDATOR scores were higher and indicative of fair condition in all subsequent sampling years, with the mean downstream O/E scores in 2009, 2011, and 2012 all significantly greater than in 2005, and not significantly different from each other from 2009 through 2013 (Figure 1B). Similarly, reversing what appeared in 2012 to be a possible downward trend in O/E scores among upstream sampling sites, with some individual sites that ranked as fair in 2005 and/or 2009 falling to poor, the mean O/E score among upstream reaches in 2013 was higher than either 2011 or 2012, and there was no significant difference among mean O/E scores at upstream sites between any of the five sampling years (Figure 1B).

Figure 1. Whychus Creek sample site PREDATOR scores (Western Cordillera + Columbia Plateau, p_c>0.5)

A. PREDATOR O/E scores across all sampling sites.



B. Changes in PREDATOR O/E scores in different stream reaches. Letter pairs indicate significant difference between mean values (P<0.05). Downstream = RM 0.5 – 9.5; mid-reach = RM 18.0 – 19.5; upstream = RM 23.5 – 30.25.



In contrast, mean PREDATOR score among sites in the midstream reaches (RM 18-19.5) showed more variation, with a mean O/E score indicating good conditions in 2005 and 2009 dropping significantly in 2011 to a score indicating poor conditions. The mean O/E score among mid-reach sampling sites was higher in 2012 than in 2011 (although not significantly) and was near the transition between poor and fair, but in 2013 the mean O/E score was almost the same as in 2011. PREDATOR scores in the mid-stream reaches may be expected to be in flux, as sampling sites include reaches at and around the newly restored Camp Polk Meadow site. The stream was re-directed into the new meandering channel in February 2012 and conditions are still changing, so mid-reach PREDATOR scores will probably continue to fluctuate over the next few years. Even a cursory visual inspection of the newly restored area reflects substantial changes in the past two years. In August 2012, a few months after the stream had been restored to a new meandering channel at Camp Polk, the old channel still had substantial flow; the substrate had a high density of black fly larvae (Simuliidae), which are characteristic of flowing water, and *Hydra*, which are not uncommon in mountain streams, were also abundant. By August 2013, most of the flow had been diverted and the old channel contained fauna much more characteristic of still waters, including deerfly (Tabanidae) larvae, toad bugs (*Gelastocoris*), and pond snails (*Physa*). Accompanying changes are occurring in the new channel, which also seemed to be undergoing some areas of erosion in 2013.

Missing and replacement taxa

PREDATOR creates a matrix showing the probability of capture of each taxon at each sampling site and the number of sites where that taxon was actually found. Some taxa that are expected to occur may be absent (missing), while others may be present at a greater number of sampling sites than predicted (replacement). There was a great deal of similarity among missing taxa from 2005 to 2013 when taxa absent from ≥7 sites are considered. In 2005 and 2009, missing taxa were almost identical, and included *Epeorus* (a sensitive flatheaded mayfly genus), *Calineuria* (a moderately sensitive perlid stonefly genus), Tanypodinae (a common non-biting midge group), Pisidiidae (common and widespread fingernail clams), and *Malenka* (a common small brown stonefly); additionally, in 2005 Leptophlebiidae (a moderately sensitive prong gill mayfly family) were absent from ≥7 of the sites where they were expected to occur. Expected taxa missing from ≥7 sampling sites in 2011 included most of the above (Leptophlebiidae, *Malenka*, Pisidiidae, Tanypodinae, and *Calineuria*), plus Chironomidae (a family of common non-biting midges, of which Tanypodinae is a subfamily), *Optioservus* and *Zaitzevia* (tolerant riffle beetle genera), and *Hydropsyche* (a tolerant net-spinning caddisfly). In 2012, the number of taxa missing from ≥7 expected sampling sites was a more restricted subset of that seen in 2011, consisting only of *Malenka*, Tanypodinae, Pisidiidae, *Calineuria*, and *Hydropsyche*, and Chironominae. Chironominae, the only missing taxon in 2012 that had not been missing in the previous years, is a subfamily of the non-biting midges (Chironomidae), which had been among the missing taxa groups in 2011. The majority of missing taxa in 2013 were identical to those seen in previous years (Leptophlebiidae, *Malenka*, Pisidiidae, Tanypodinae, *Calineuria*, *Epeorus*, *Hydropsyche*), with the addition of three taxa that had never been noted before as missing: *Micrasema*, a genus of humpless case maker caddisfly (Brachycentridae) that is found in areas of moderate to fast current and is sensitive to pollution and human disturbance; *Neophylax*, a genus of pollution-sensitive stonecase (Uenoidae) caddisfly; and Ceratopogoninae, a moderately tolerant subfamily of biting midge.

Substantial similarity was also seen among replacement taxa across the five years of sampling. Diamesinae (a non-biting midge group), *Serratella* (a commonly-collected genus of spiny crawler mayfly), *Rhithrogena* (a common and abundant flatheaded mayfly genus), *Acentrella* (a common small minnow mayfly genus), *Narpus* (a common, moderately tolerant riffle beetle genus), and *Atherix* (a common, tolerant watersnipe fly genus) were present as replacement taxa in all years. Additional replacement taxa seen in 2011 included Nematoda (common roundworms), Turbellaria (flatworms), *Drunella* (a sensitive spiny crawler mayfly genus), *Rickera* (a sensitive stripetail stonefly genus), and *Brachycentrus* (a sensitive genus of humpless case-making caddisfly). The number of replacement taxa was higher in 2012 than in previous years and differed more in the over-represented groups; in addition to the taxa mentioned above as replacements in all years, Capniidae (a sensitive family of stoneflies), *Suwallia* (a sensitive genus of chloroperlid stonefly), *Agapetus* (a sensitive genus of saddlecase-making caddisfly), *Brachycentrus* (also a replacement taxon in 2011), and *Neoplasta* (a moderately tolerant genus of dance fly) occurred as replacement taxa. This may suggest a period of greater dynamism in the macroinvertebrate community, which would not be unexpected, as the creek was moved to its restored channel in Camp Polk Meadow only six months prior to the 2012 sampling. Replacement taxa in 2013 samples were similar to those in previous years, and included 14 taxa that were also replacements in 2012 samples. Two taxa that had never been seen in previous years in high enough numbers to qualify as replacement taxa were found as replacements in the 2013 samples: *Ephemerella* and *Attenella*, two different genera of sensitive mayflies in the spiny crawler family (Ephemerellidae).

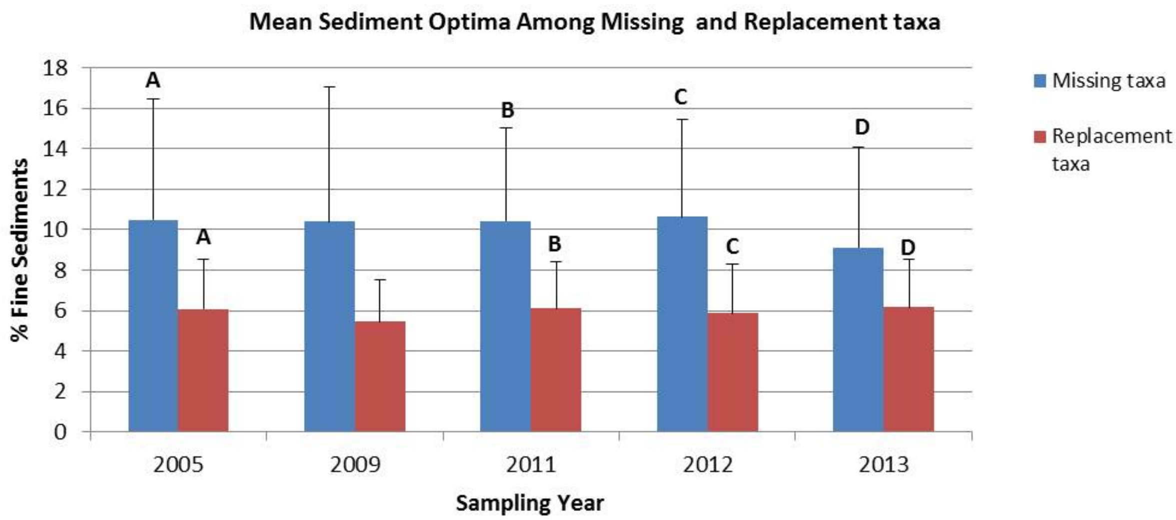
Sediment and temperature optima

Differences in tolerance for sediments and temperature among missing and replacement taxa were examined to investigate whether either factor is acting as a stressor, using a set of optima values for specific macroinvertebrate taxa for seasonal maximum temperature and percent fine sediments developed by ODEQ (Huff *et al*. 2006). These comparisons indicate a strong and consistent difference between fine sediment tolerances among missing and replacement taxa, but show that temperature tolerances are quite similar between the two groups (Figure 2).

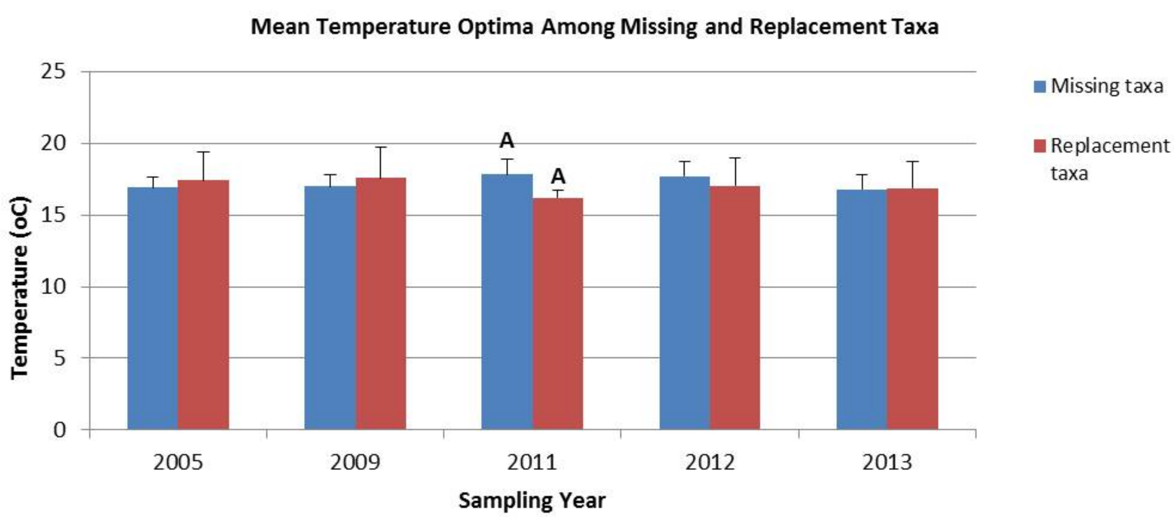
The mean sediment optima for replacement taxa was substantially and significantly lower than that of missing taxa across all years except in 2009 (though the difference between missing and replacement taxa in that year approached significance, at $p = 0.0559$). In contrast, a significant difference in temperature optima between missing and replacement taxa was seen only in 2011. In-stream restoration work such as channel reconstruction can alter substrate mobility and heterogeneity and affect invertebrate community composition (Albertson *et al*. 2011). Multiple restoration projects done in Whychus Creek to increase flow and restore the stream to its natural channel would be expected to affect both temperature and sediment characteristics, but sediment is a much more strong and consistent driver of missing vs. replacement taxa.

Figure 2. Comparison of temperature and sediment optima among missing and replacement taxa

A. Mean sediment optima of missing vs. replacement taxa. Letters indicate significant difference between mean values (P<0.05). Note that in 2009, the difference between the optima was close to significant (p = 0.0559).



B. Mean temperature optima of missing vs. replacement taxa. Letter pairs indicate significant difference between mean values (P<0.05).



Fine sediment is frequently a significant source of nonpoint pollution and habitat impairment in streams. Increased fine sediment loads have a negative impact on endangered salmonids (Waters 1995, Soulsby et al. 2001, Suttle et al. 2004) and can disrupt macroinvertebrate feeding, respiration, and mobility. In this case, however, the replacement taxa assemblages have much lower tolerance for fine sediments than the missing taxa assemblages, suggesting lower than expected sediment levels. While sediment flow is affected by both stream power and sediment supply, the flow rate of most rivers is unlikely to be a limiting factor in the capacity to transport fine sediments. One possible explanation for the low sediment optima observed for replacement taxa is that sediment supply is limited. Another explanation is that the Whychus system may naturally be characterized by lower sediment than would be expected according to the taxa considered to be replacement taxa by the PREDATOR model. Because fine sediment can be washed out of riffles in streams with high flow (Jackson & Beschta 1982, Lisle & Hilton 1992), and the DEQ sampling protocol is done only in riffle habitats, the differences in assemblage sediment optima may be even more pronounced.

Multimetric assessment

The OWEB Level 3 stream IBI (genus/species) has 10 metrics: Taxa richness (# of taxa at site); Ephemeroptera (mayfly) richness; Plecoptera (stonefly) richness; Trichoptera (caddisfly) richness; # sensitive taxa; # sediment-sensitive taxa; % dominance of the top taxon; % tolerant taxa; % sediment-tolerant taxa; and modified Hilsenhoff Biotic Index (MHBI). The raw value of each metric is calculated and accorded a corresponding scaled score of 5, 3, or 1, with higher scores indicating conditions closer to reference. Scaled values for individual metrics are summed to yield a single IBI score for each site, which can reflect minimal (>39), slight (30-39), moderate (20-29), or severe biological impairment (<20).

IBI-based assessment presents a slightly better picture of stream biotic health than PREDATOR (Table 3; Figure 3). IBI scores overall indicate better biotic conditions than O/E scores for the same sites, although sites that received higher IBI scores also generally received higher O/E scores. No site ever scored as severely impaired, and in every year except 2009, at least one site scored as being minimally disturbed (Table 3; see also Appendix C for individual metric values and scaled IBI scores at each site).

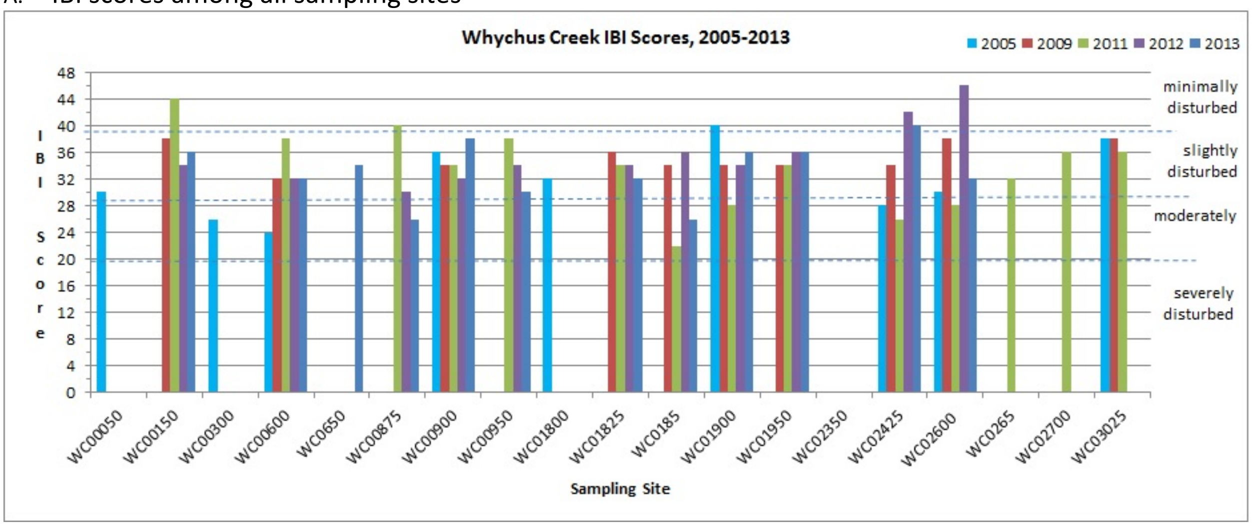
Table 3. IBI Scores. Colors indicate minimal (orange), slight (green), or moderate disturbance (blue). (Note that no site received a score in any year indicating severe disturbance).

Site	2005	2009	2011	2012	2013
WC00050	30	N/A	N/A	N/A	N/A
WC00150	N/A	38	44	34	36
WC00300	26	N/A	N/A	N/A	N/A
WC00600	24	32	38	32	32
WC00650	N/A	N/A	N/A	N/A	34
WC00875	N/A	N/A	40	30	26
WC00900	36	34	34	32	38
WC00950	N/A	N/A	38	34	30
WC01800	32	N/A	N/A	N/A	N/A
WC01825	N/A	36	34	34	32
WC01850	N/A	34	22	36	26
WC01900	40	34	28	34	36
WC01950	N/A	34	34	36	36
WC02425	28	34	26	42	40
WC02600	30	38	28	46	32
WC02650	N/A	N/A	32	N/A	N/A
WC02700	N/A	N/A	36	N/A	N/A
WC03025	38	38	36	N/A	N/A

IBI scores were similar to PREDATOR scores in that the mean in downstream reaches across all years reflected a more sustained improvement and stability in biological condition. The mean IBI score among downstream sites was significantly lower in 2012 and 2013 compared to 2011, but the overall condition each year was still slightly disturbed (Figure 3B). In contrast to the significant fluctuation seen in mean PREDATOR scores for sites in the mid-stream reaches, mean mid-reach IBI scores were more stable; overall mean scores differed between years, but not significantly, and all indicated slightly disturbed conditions, with the exception of 2011, when the mean IBI score among mid-reach sites was at the transition between moderately and slightly disturbed (mean score = 29.5) . Mean IBI scores for the upstream sampling sites (RM 23.5 to 30.25) were significantly higher in 2012 compared to 2005 and 2011, with a mean value reflecting minimally disturbed conditions in 2012; this value dropped again in 2013, although the difference was not significant.

Figure 3. Level 3 IBI scores for Whychus Creek sites

A. IBI scores among all sampling sites



B. Changes in IBI scores based on stream reach. Letter pairs indicate significant difference between mean values (p<0.05). Downstream reach = RM 0.5 – 9.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

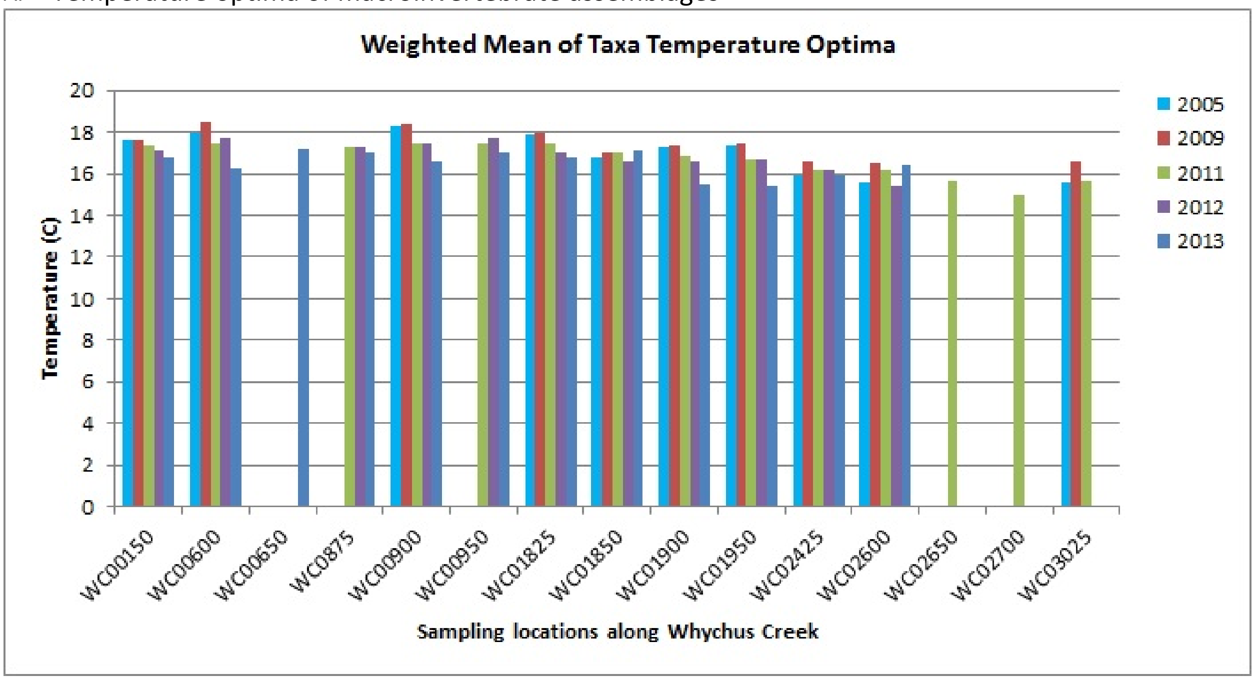
Macroinvertebrate Community Composition

Temperature and Fine Sediment Optima

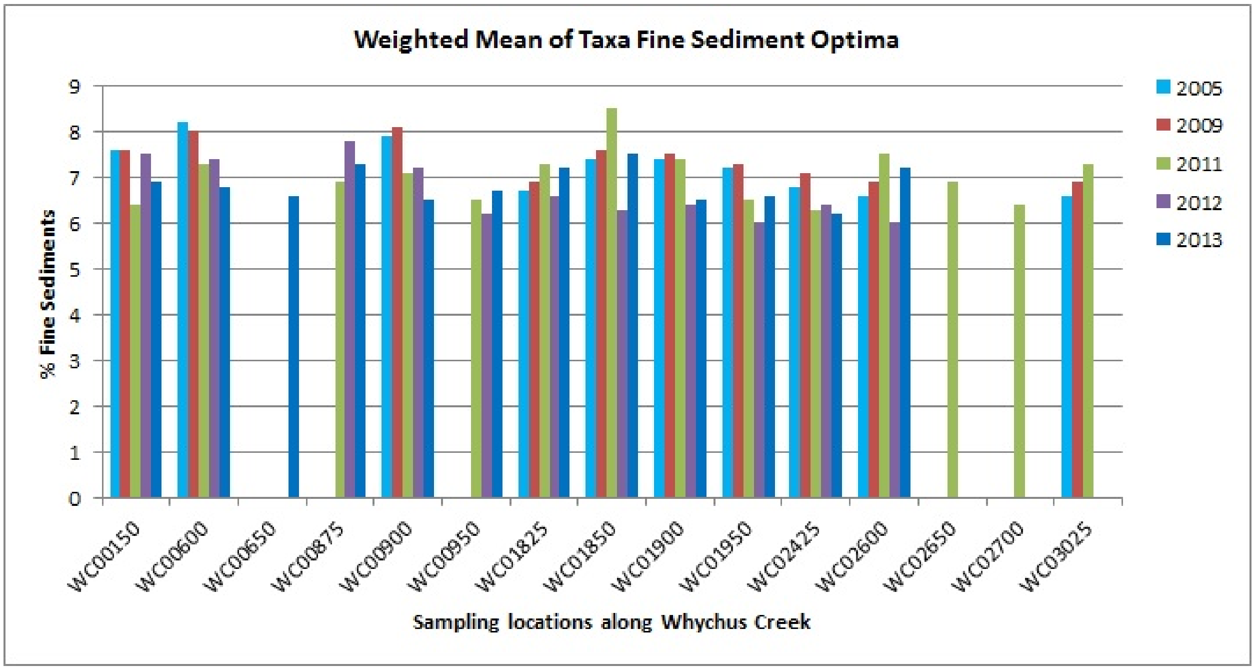
The mean optima values for temperature and fine sediments among the macroinvertebrate assemblages at each site over time were determined (Figure 4). The communities differed in both temperature and sediment optima from site to site, though the magnitude of change differed among sites, and varied more annually at each site for sediment than temperature.

Figure 4. Temperature and fine sediment optima of macroinvertebrate assemblages at sampling sites

A. Temperature optima of macroinvertebrate assemblages



B. Fine sediment optima of macroinvertebrate assemblages

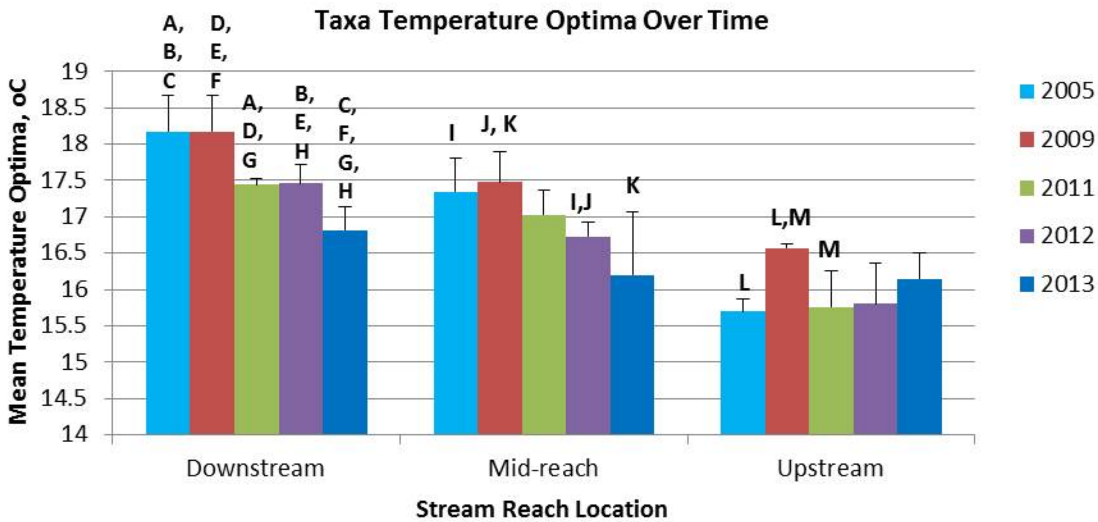


When viewed at the reach level, substantial differences are seen in mean optima values among the communities at downstream, mid-reach, and upstream sites, and significant changes in assemblage optima in individual reaches are apparent (Figure 5). The mean temperature optima of macroinvertebrate assemblages differs based on overall site location, with upstream sampling sites having the lowest overall temperature optima and downstream sites the highest. This is to be expected based on changes in stream conditions and biota as predicted by the River Continuum Concept (Vannote *et al*. 1980) and observable in the stream itself, as the colder, faster, deeper headwater regions transition to shallower stretches flowing through areas with a higher degree of human disturbance and impervious surface in the landscape. In addition, a significant downward trend in temperature assemblage optima was seen from 2005 to 2011 at all three reaches (Figure 5A). This shift was thought to have stabilized by 2012, as there was no significant difference in mean temperature optima between macroinvertebrate assemblages collected in 2011 and 2012 at sampling sites within the three different reaches, but in 2013, the mean temperature optima of the macroinvertebrate assemblage was again significantly lower at downstream and mid-reach sites. In contrast, the mean temperature optima of the upstream reach assemblages in 2012 and 2013 were not significantly different from the 2011 mean.

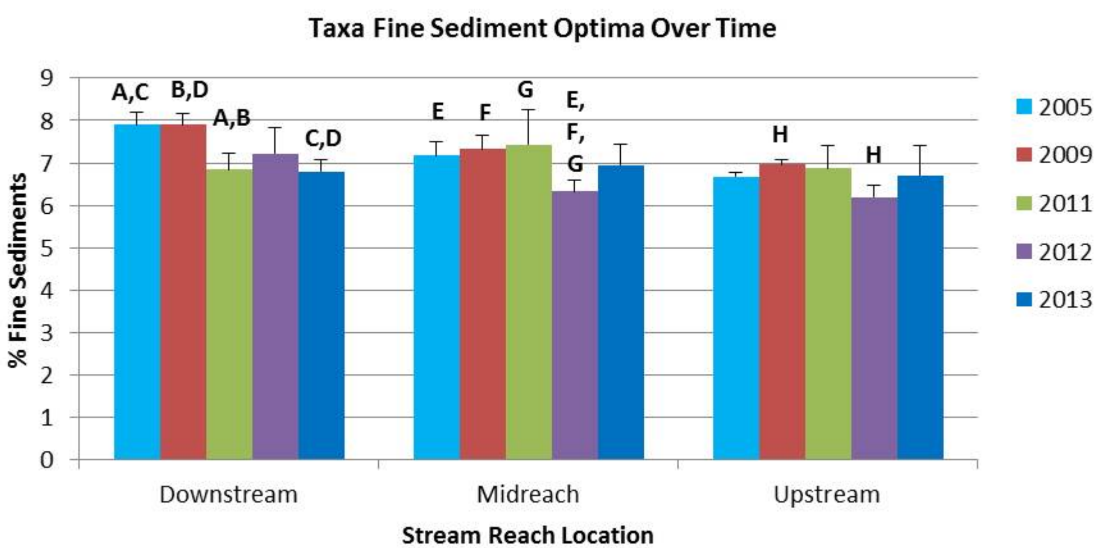
A less pronounced pattern was seen for changes in mean sediment optima among communities within different reaches (Figure 5B). A significant decrease in mean assemblage sediment optima occurred among downstream reaches through 2013, but among mid-reach and upstream sites, although the mean optima for assemblages in 2012 were significantly lower than in one or more previous years, there was no significant difference in assemblage sediment optima in 2013, although the means were slightly higher.

Figure 5. Changes in macroinvertebrate assemblage temperature and fine sediment optima over time. Letter pairs indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 9.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

A. Changes in macroinvertebrate assemblage optima over time



B. Changes in macroinvertebrate assemblage fine sediment optima over time



EPT

Mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) are considered the most sensitive of aquatic macroinvertebrate groups and are often used as a measure of stream biological quality. Most taxa in these orders require cold, clean, well-oxygenated water, and a greater abundance and diversity of EPT taxa is correlated with better stream conditions. The number of EPT taxa collected among all sampling sites (richness) increased slightly from 2005 through 2011 (Table 4), but the proportion of all taxa collected comprised by EPT remained roughly the same, with the proportion of total taxa comprised by Trichoptera and Ephemeroptera consistently higher than Plecoptera.

Table 4. EPT composition of macroinvertebrate community. Numbers in parentheses indicate proportion of total taxa collected in each year comprised by each group.

Year	Total # of taxa	Total #EPT	# Ephemeroptera taxa	# Plecoptera taxa	# Trichoptera taxa
2005	76	42 (55.3%)	14 (18.4%)	11 (14.5%)	17 (22.4%)
2009	85	47 (55.3%)	14 (16.5%)	13 (15.3%)	20 (23.5%)
2011	82	49 (59.8%)	17 (20.7%)	13 (15.9%)	19 (23.2%)
2012	79	44 (55.7%)	19 (24.1%)	11 (13.9%)	14 (17.7%)
2013	83	44 (53%)	17 (20.5%)	10 (12%)	17 (20.5%)

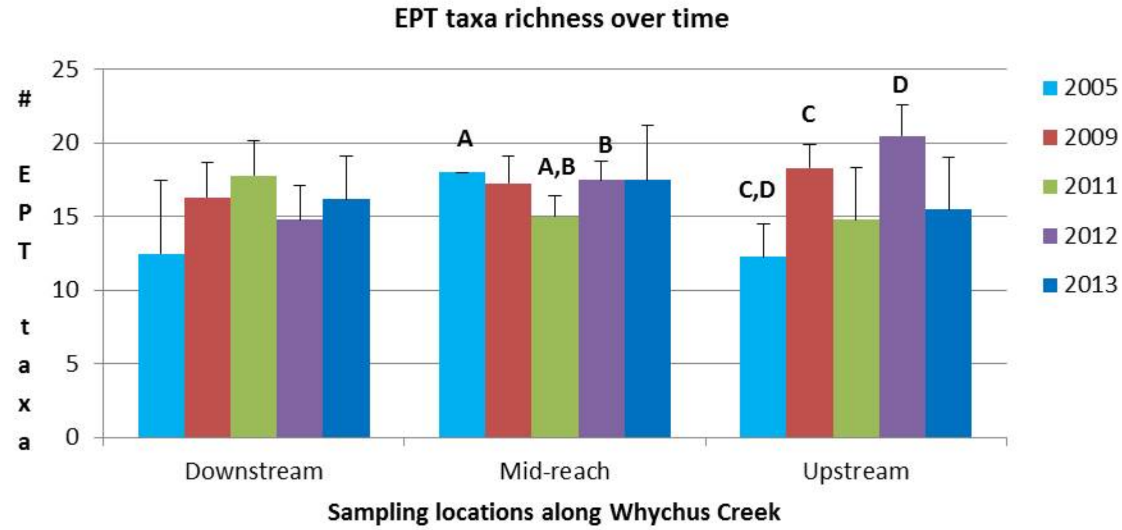
Because macroinvertebrates respond to microhabitat features that differ among stream reaches, change in EPT composition across sampling years was examined for sites in the downstream (RM 0.5-9.5), mid-reach (RM 18-19.5), and upstream (RM 23.5-30.25) portions of the creek (Figure 6). The downstream reaches showed a trend toward increased number of EPT taxa, relative proportion of EPT taxa, and relative abundance of EPT from 2005 to 2009 which then stabilized through 2013, although only the mean relative abundance of EPT among downstream sites was significantly greater in 2011-2013 compared to 2009 (Figure 6C). Among mid-reach sites, the mean number, proportion of taxa, and relative abundance of EPT had increased significantly by 2012; in 2013, mean EPT taxa richness among mid-reach sites remained essentially the same as in 2012, but both relative EPT richness and relative abundance decreased, though the difference was significant only for relative EPT abundance.

The most sustained and dramatic change in EPT composition was seen among upstream sampling sites, with EPT richness, diversity and relative abundance all trending steadily and significantly upwards from 2005 through 2012. However, in 2013 the mean values for all three EPT metrics had decreased, although the difference from 2012 values was significant only for relative EPT richness.

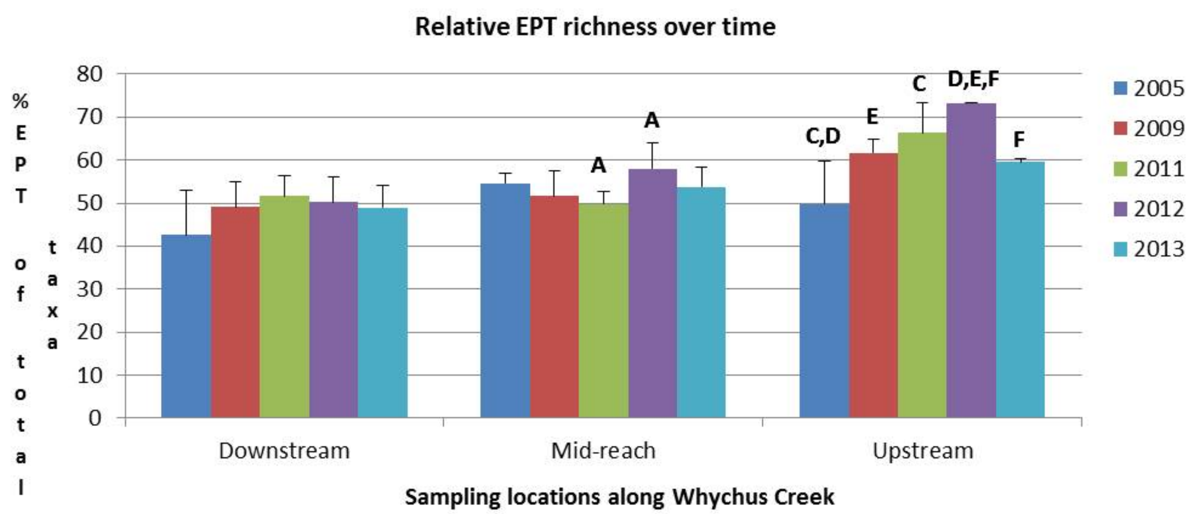
The substantial and significant decreases in relative EPT abundance and richness at mid-stream and upstream sites and in EPT richness at upstream sites from 2012 to 2013 may indicate changes in stream conditions associated with the Pole Creek fire, which occurred in the headwaters of Whychus Creek in September, 2012.

Figure 6. Changes in EPT taxa. Letters indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 9.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

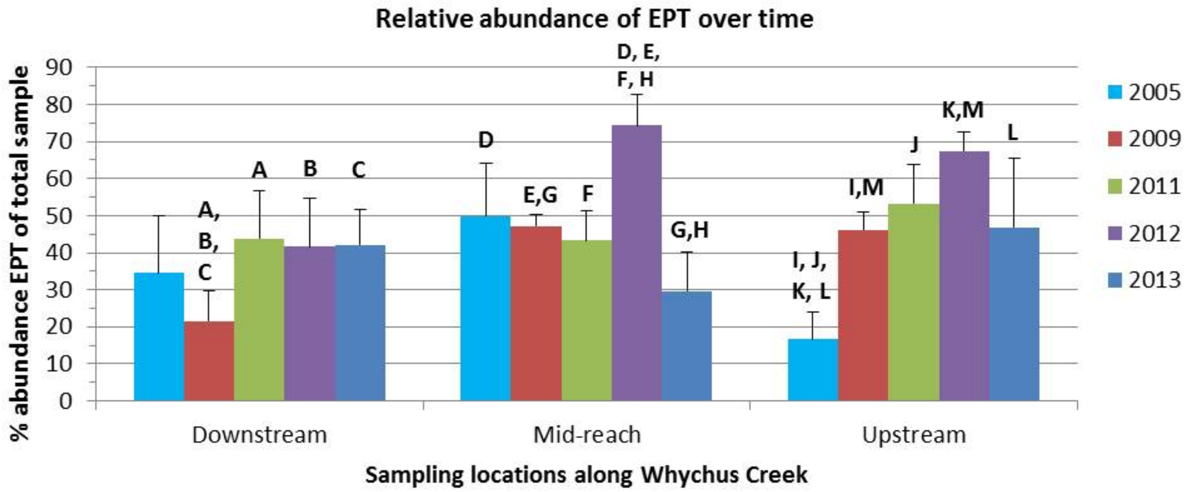
A. EPT richness (mean of number of different EPT taxa collected)



B. Relative EPT richness (mean of # EPT taxa / total # taxa at each site)



C. Relative abundance of EPT (# individuals in EPT / total # organisms per site)



Community composition

In 2005 and 2009, many taxa were found at only a single site (29 taxa and 30 taxa, respectively) and generally with anywhere from one to four individuals. This number was substantially lower in 2011, when only 13 of the 82 total taxa were collected from just one site, also at very low abundance. Similarly in 2012, 18 of the 79 taxa collected were found at only a single site, with no more than one to two individuals, and in 2013, 20 of the 83 taxa collected occurred at only a single site. Of the 133 total taxa collected during this project (2005-2013), 35 were present in all five sampling years (see Appendix D for a complete taxa list). These were comprised primarily of riffle beetles (Elmidae), midges (Chironomidae), blackflies (*Simulium*), and small minnow mayflies (Baetidae), all common taxa expected to occur widely and the most abundant and ubiquitous in this study.

There were some instances where a difference was seen in the presence or absence of a certain group in 2005 compared to later sampling years. *Rhyacophila* caddisflies were absent from all samples in 2005 but present as multiple different species in 2009-2013, especially in the mid- to upstream reaches of the creek; similarly, the riffle beetle *Lara avara* and several species of *Epeorus* mayflies were absent in 2005 but present in later years. These groups are more sensitive and prefer cooler, faster-flowing water. In contrast, several genera of caddisflies in the family Limnephilidae (Northern caddisflies) were present among sites in 2005 but absent from samples in later years, and coenagrionid damselfly nymphs were also seen only in 2005. Coenagrionids are a tolerant damselfly family that prefers to breed in slower waters, and limnephilids can often be found in more lentic (slow-flowing) and/or temporary waters. Thus, post-restoration changes in stream flow and temperatures likely influenced this community shift.

Analysis of a Bray-Curtis similarity matrix of square root-transformed abundance data suggests a greater overall change in community composition from 2005 and 2009 compared to later years. Among the five sites that were sampled in all five years (WC00600, WC00900, WC01900, WC02425, WC02600), mean macroinvertebrate community similarity decreased when the community present in 2005 was compared to that of every subsequent year at the same site. Community composition among all sites sampled across four years from 2009-2013 (WC00150, WC01825, WC01850, WC01950) was consistently greater and changed less from year to year.

A shift in macroinvertebrate community composition followed by greater stability is further suggested by CLUSTER analysis and MDS ordination (PRIMER v6), which grouped all 2005 samples separately from all samples taken in subsequent years, with the exception of the anomalous duplicate sample taken at RM 24.25 in 2013, with an average similarity between the 2005 sample cluster and the 2009-2013 cluster of only 30.6% (Figure 7A). Samples from other years clustered together based on stream reach location. Overall similarity of the communities within each reach group was similar, with all upstream sites from 2009-2013 clustering together at 50% similarity, those from all mid-stream reaches clustering at 51% similarity, and those from all downstream sampling sites in 2009-2013 clustering together at 54% community similarity (Figure 7).

MDS ordination illustrates the difference between 2005 and all other sampling years even more dramatically (Figure 8), with the 2005 samples well-separated from all other years. MDS ordination further shows that sample clustering is influenced strongly by site location, with samples taken from similar reaches of the creek (upstream, mid-reach, or downstream) exhibiting the greatest similarity between years, although the samples still tend to cluster more closely with those taken from the same reach area.

Replicate samples (DUP) taken in all four years for quality assurance purposes clustered most closely with each other in each year, with the exception of one replicate sample in 2012 (WC00600) that clustered most closely with the sample taken at the same site the previous year, and an anomalous sample taken in 2013 at RM 24.25. This indicates that the sampling technique and training provided were sufficiently standardized that volunteers in each year obtained similar results when taking replicate samples in the eight riffles of a selected stream reach, which is a consideration when the volunteers changed with each year, and were usually completely new to the techniques used.

Figure 7. CLUSTER analysis of macroinvertebrate community composition data from 2005-2013. The final two digits of each sampling site label refer to year. DUP indicates a replicate sample taken for quality control purposes.

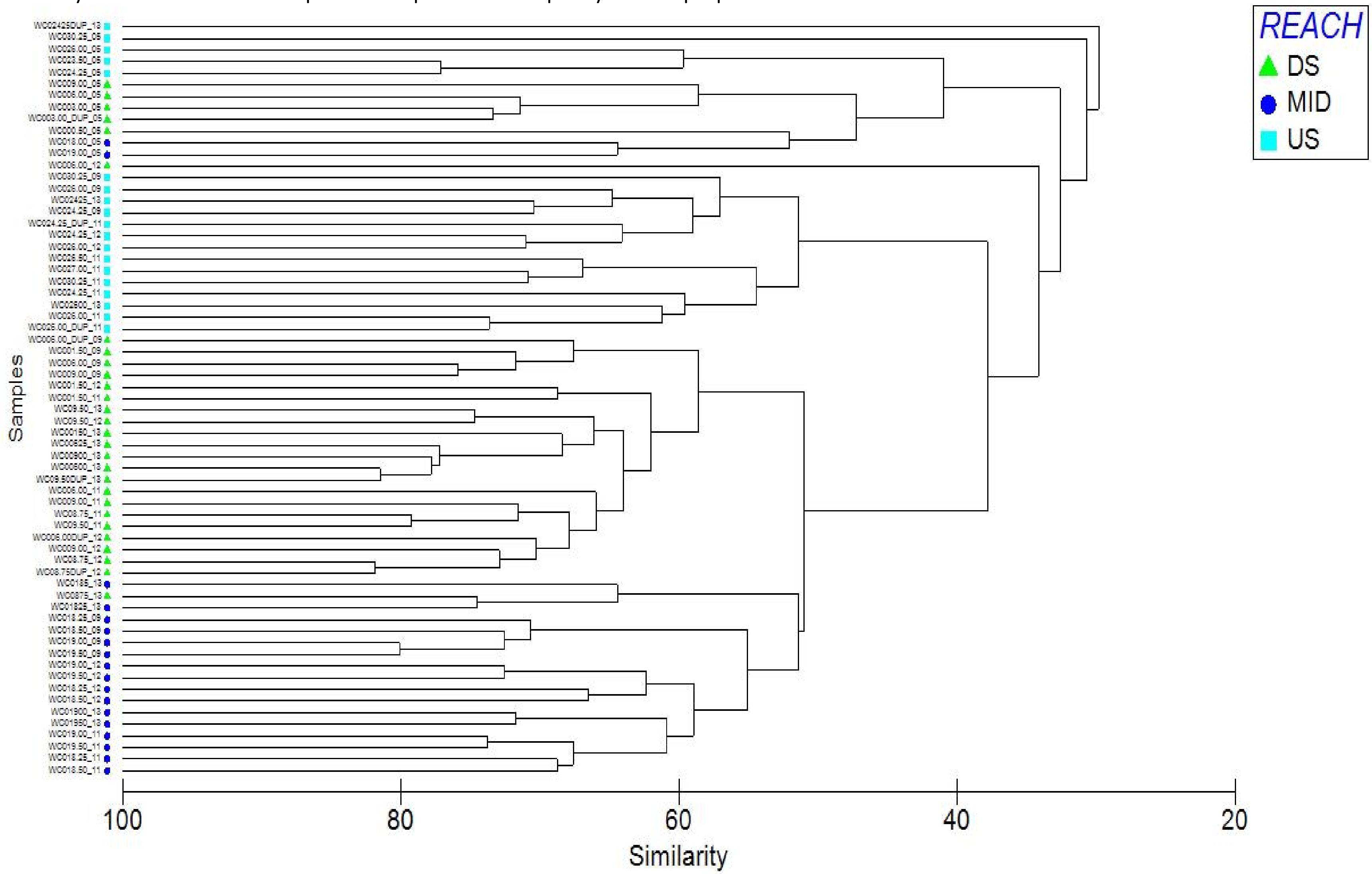
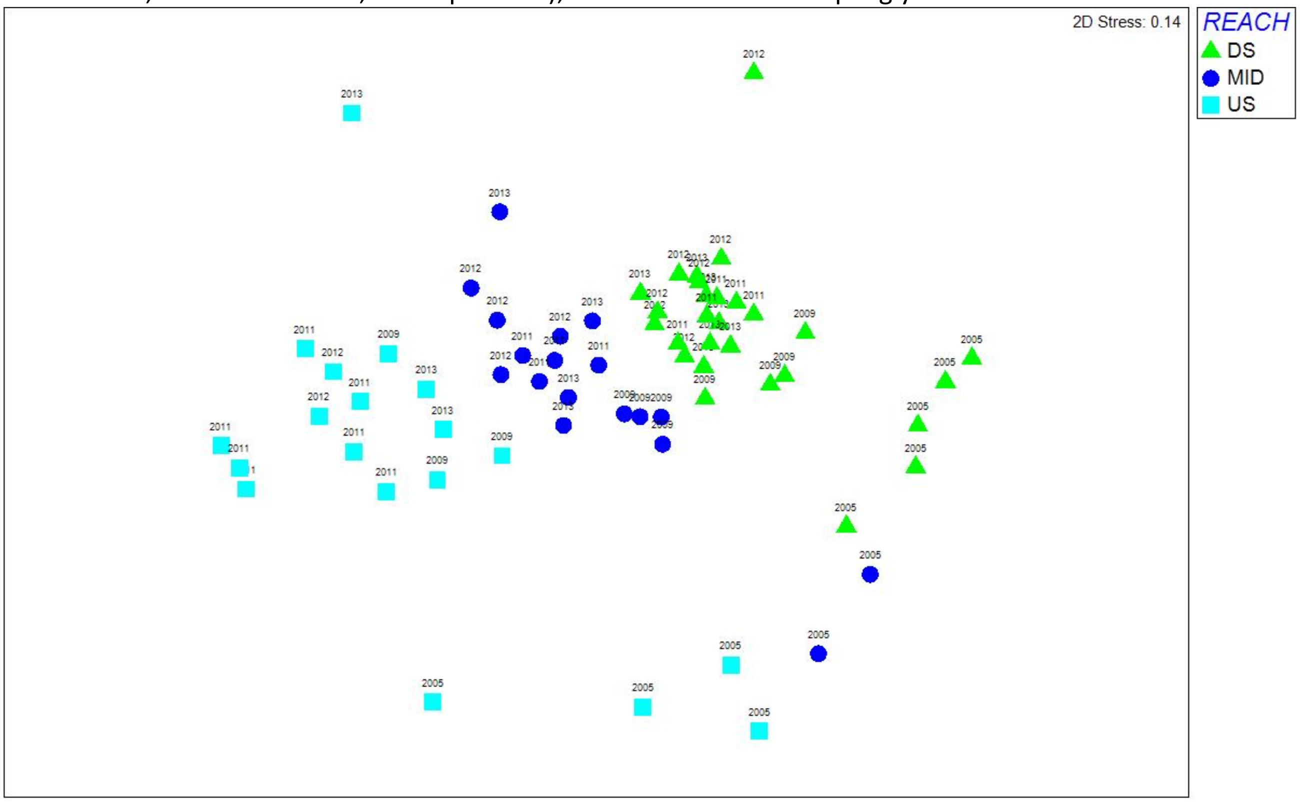


Figure 8. MDS ordination of macroinvertebrate assemblages. Symbols indicate stream reach (DS = downstream, MID = mid-reach, US = upstream); number indicates sampling year.



Conclusions

Many restoration projects are undertaken with the assumption that creating improved physical habitat automatically leads to increased biodiversity, which in turn restores impaired or lost ecological processes. This "field of dreams hypothesis" (Palmer *et al*. 1997) has not always been borne out in practice, and a variety of reach- and catchment-specific influences must be considered when evaluating project outcomes (Roni *et al.* 2002, Bond & Lake 2003, Palmer & Allan 2006, Lake *et al*. 2007, Ernst *et al*. 2012, Haase *et al*. 2013). Restoration activities can improve habitat and water quality at the reach level, but streams experience significant watershed-wide stressors that act as limiting factors and may not be completely remediated by site-specific activities (Bohn & Kershner 2002, Bond & Lake 2003, Bernhardt & Palmer 2011, Kail *et al*. 2012). Distribution of aquatic biota within a stream is additionally governed by hydrologic effects such as velocity, turbulence, suspended load, and bedload movement (Gore *et al*. 2001).

Given these caveats, it is essential that effectiveness monitoring be done on a regular basis over multiple years. Stream degradation occurs over the long-term, and by the same token, recovery of stream biota is not instantaneous. As stream habitat improves, new individuals can be recruited into the area, but the time frame needed for re-colonization and establishment of stable reproducing populations will vary for different taxa with different life histories and colonization abilities (Niemi *et al*. 1990), and may be longer than the time frame of many monitoring programs. In addition, unlike a "pulse"-type disturbance such as a flood, "press" disturbances such as restoration activities take place over a longer period of time, and recovery rates of aquatic biota are generally longer than expected for press disturbances as opposed to pulses (Yount & Niemi 1990). The rate of colonization and ultimate taxa composition is also strongly affected by the available source populations in the region and their ability to disperse (Brederveld *et al*. 2011). Only by taking regular "snapshots" of one or more selected biotic communities is it possible to know if the macroinvertebrate community is still changing or if it has stabilized, and whether suspected trends reflect true restoration-related changes in the biotic community or are the isolated result of anomalous conditions in a single year.

Aquatic invertebrate monitoring in Whychus Creek has been done five times within the past eight years, a timeframe that allowed baseline conditions to be established and subsequent post-restoration changes to be identified and assessed. Analysis of macroinvertebrate community data collected from 2005-2013 indicates the following:

- The overall composition of the benthic macroinvertebrate community in Whychus Creek changed substantially from 2005 to 2009, but has shown increasing stabilization since. CLUSTER and MDS ordination show the greatest change in macroinvertebrate community composition occurred between 2005 and 2009, with community composition among each subsequent pair of years becoming increasingly similar. This trend appears to illustrate a real shift in the macroinvertebrate community between 2005 and 2009 compared to 2009-2013, rather than an artifact of sampling frequency, given that 2009-2013 points are intermingled in the MDS ordination and in CLUSTER analysis, with 2005 points occupying positions separate from the subsequent four data years in both analyses. This shift may represent an initial community response to restoration activities, specifically stream flow restoration, followed by recolonization and stabilization within the macroinvertebrate community. Flow restoration that dramatically increased August median flows between 2005 and 2009 may have had a relatively greater effect on macroinvertebrate populations compared to more recent restoration actions; it is also possible that additional stressors unmitigated by recent stream work are still exerting an effect on the community.

- PREDATOR scores consistently indicate lower biotic conditions than IBI scores at the same sites. Overall, PREDATOR scores suggest that sites in the downstream reach (RM 0.5 to 9.5) showed a slight improvement from 2005 to 2009 that then stabilized across the remaining sampling years. Although a downward trend in PREDATOR score was seen among upstream sites (RM 23.5 to 30.25) from 2005-2012, the change was not significant in any year, and the mean PREDATOR score among upstream sites increased again in 2013, although not significantly, due in part to a large increase in the O/E score at City Park (RM 24.25). PREDATOR scores at sites within the mid-reach region showed the greatest fluctuation over time, and are likely to continue to change as the stream responds to the recent relocation to its restored channel at Camp Polk.

- IBI scores for the downstream reach sampling sites show a pattern similar to the PREDATOR scores from this reach, with a period of improvement followed by stabilization. Fluctuation among the mid-reach sites was less pronounced, and differences in mean scores were not significant among any years. Mean IBI scores at upstream reach sites indicate a significant improvement in biotic conditions for sites in this region from 2005 through 2012; the mean score was lower in 2013, but the difference was not significant. The improvement in IBI scores among upstream sampling sites may be accounted for by changes in the sensitive EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa. While EPT richness, relative diversity, and abundance increased over time among all three reach regions, it was most dramatic among the upstream sampling sites. Abundance, richness, and life history characteristic of the EPT factor into multiple IBI metrics, and they may be the driving force behind the observed increase in IBI scores among upstream sites. Decreases in EPT metrics from 2012 to 2013 may indicate changes in stream conditions resulting from the September 2012

Pole Creek fire in the headwaters of Whychus Creek.

- The mean temperature optima of the macroinvertebrate assemblages are consistently higher at downstream sites and lowest at upstream sites, reflecting changes in hydrology and geomorphology along the course of the creek. Examination of missing and replacement taxa at each site as identified by PREDATOR shows that in every year, mean fine sediment optima of replacement taxa is significantly lower than that of the missing taxa. Differences in mean temperature optima among missing and replacement taxa were significant in only a single year, suggesting that stream sediment characteristics provide a better explanation for variations in observed vs. expected macroinvertebrate community than temperature conditions; the differences in stream sediment characteristics may also be attributed to an incomplete representation of the Whychus system by the PREDATOR model. Taxa temperature optima for the entire macroinvertebrate assemblage showed a significant downward trend from 2005 through 2011, which resumed from 2012 to 2013. This trend is consistent with temperatures in Whychus Creek decreasing from 2005 to 2013, as median August flows steadily increased through streamflow restoration efforts.

- Multiple years of effectiveness monitoring along Whychus Creek have enabled detection of longer term changes and trends. The data suggest that in many parts of the stream, the macroinvertebrate community composition appears to be stabilizing, although mid-reach sampling sites where restoration activates are still in progress are fluctuating more. While the initial 2005-2009 shift suggests some community response to changing stream conditions, possibly associated with streamflow restoration over this interval, consistently lower mean fine sediment optima observed in all years among replacement vs. missing taxa and the lack of minimally disturbed (IBI) or good (PREDATOR) scores suggest the composition of the macroinvertebrate community continues to reflect stream conditions which to some extent remain impaired or degraded. This assemblage may reflect a response to a number of influences, including legacy effects of historical land use and modifications to the stream (e.g. channelization, floodplain development, agriculture); natural disturbances such as geomorphic or high flow events and interannual variability in flow regime; and/or effects associated with restoration activities. Whatever the cause, it suggests an opportunity remains to improve stream conditions through carefully targeted restoration actions. Ongoing monitoring will provide additional insight into macroinvertebrate response to changing stream conditions as restoration continues.

References

Adams J., M. Vaughan, and S. H. Black. 2003. Stream bugs as biomonitors: guide to Pacific Northwest macroinvertebrate monitoring (CD-ROM). The Xerces Society for Invertebrate Conservation, Portland OR.

Albertson L. K., B. J. Cardinale, S. C. Zeug, L. R. Harrison, H. S. Lenihan, and M. A. Wydzga. 2011. Impacts of channel reconstruction on invertebrate assemblages in a restored river. Restoration Ecology 19(5): 627-638.

Barbour M.T., J. B. Stribling, J. Gerritsen, and J. R. Karr. 1996. Biological criteria: technical guidance for streams and small rivers. EPA 822-B-96-001, United States Environmental Protection Agency, Office of Water, Washington DC.

Bernhardt, E.S. and M.P. Palmer. 2011. River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. Ecological Applications 21(6): 1926-1931.

Bohn B. A. and J. L. Kershner. 2002. Establishing aquatic restoration priorities using a watershed approach. Journal of Environmental Management 64: 355-363.

Bond N. R. and P. S. Lake. 2003. Local habitat restoration in streams: constraints on the effectiveness of restoration for stream biota. Ecological Management & Restoration 4(3): 193-198.

Brederveld, R.J., S.C. Jahnig, A.W. Lorenz, S. Brunzel, and M.B. Sooms. 2011. Dispersal as a limiting factor in the colonization of restored mountain streams by plants and macroinvertebrates. Journal of Applied Ecology 48: 1241-1250.

Clarke K. R. and R. M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd ed. PRIMER-E, Plymouth, United Kingdom.

Connell J. H. 1978. Diversity in tropical rain forests and coral reefs. Science 199: 1302–1310

Crunkilton, R. L. and R. M. Duchrow. 1991. Use of stream order and biological indices to assess water quality in the Osage and Black river basins of Missouri. Hydrobiologia 224: 155-166.

Ernst, A.G., D.R. Warren, and B.P. Baldigo. 2012. Natural-channel-design restorations that changed geomorphology have little effect on macroinvertebrate communities in headwater streams. Restoration Ecology 20(4): 532-540.

Gibson R. J. 1993. The Atlantic salmon in fresh water: spawning, rearing, and production. Reviews in Fish Biology and Fisheries 3: 39-73.

Gore J. A., J. B. Layzer, and J. Mead. 2001. Macroinvertebrate instream flow studies after 20 years: a role in stream management and restoration. Regulated Rivers Research & Management 17: 527-542.

Haase, P., D. Hering, S.C. Jahnig, A.W. Lorenz and A. Sunderman. 2013. The impact of hydromorphological restoration on river ecological status: a comparison of fish, benthic invertebrates, and macrophytes. Hydrobiologia 704: 475-488.

Hilsenhoff W.L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20:31-39.

Hubler S. 2008. PREDATOR: development and use of RIVPACS-type macroinvertebrate models to assess the biotic condition of wadeable Oregon streams. DEQ08-LAB-0048-TR, State of Oregon Department of Environmental Quality, Laboratory Division, Watershed Assessment Section. 51 pp.

Huff D. D., S. Hubler, Y. Pan, and D. Drake. 2006. Detecting shifts in macroinvertebrate community requirements: Implicating causes of impairment in streams. DEQ06-LAB-0068-TR. State of Oregon Department of Environmental Quality, Laboratory Division, Watershed Assessment Section.

Jackson, W. L. and R. L. Beschta. 1982. A model of two-phase bedload transport in an Oregon Coast Range stream. Earth Surface Processes and Landforms 7(6): 517-527.

Kail, J., J. Arle, and S. J. Jahnig. 2012. Limiting factors and thresholds for macroinvertebrate assemblages in European rivers: empirical evidence from three datasets on water quality, catchment urbanization, and river restoration. Ecological Indicators 18: 63-72.

Karr J.R. and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington D. C. 206 pp.

Lake P. S., N. Bond, and P. Reich. 2007 Linking ecological theory with stream restoration. Freshwater Biology 52: 597–615.

Lillie, R. A., S. W. Szczytko, and M. A. Miller. 2003. Macroinvertebrate data interpretation guidance manual. Wisconsin Department of natural Resources, Madison WI. 64 pp.

Lisle, T. E. and S. Hilton. 1992. The volume of fine sediment in pools: an index of sediment supply in gravel-bed streams. Water Resources Bulletin 28(2): 371-383.

Nedeau E. J., A. K. Smith, J. Stone, and S. Jepsen. 2009. Freshwater mussels of the Pacific Northwest, 2nd ed. The Xerces Society for Invertebrate Conservation, Portland OR. 51 pp.

Niemi, G. J., P. DeVore, N. Detenbeck, D. Taylor, A. Lima, and J. Pastor. 1990. Overview of case studies on recovery of aquatic systems from disturbance. Environmental Management 14(5): 571–587.

Oregon Watershed Enhancement Board. 2003. OWEB water quality monitoring technical guidebook. 152 pp.

Palmer M. A. and J. D. Allan. 2006. Restoring rivers. Issues in Science and Technology pp. 40-48.

Palmer M. A., R. F. Ambrose, and L. N. Poff. 1997. Ecological theory and community restoration ecology. Restoration Ecology 5: 291–300.

Plafkin J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. *Rapid bioassessment protocols for use in streams and rivers: Benthic macroinvertebrates and fish.* U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, D.C. EPA 440-4-89-001.

Roni P., T. J. Beechie, R. E. Bilby, F. E. Leonetti, M. M. Pollock and G. R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific northwest watersheds. North American Journal of Fisheries Management 22: 1-20.

Rosenberg D. M. and V. H. Resh. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York, NY.

Soulsby, C., A. F. Youngson, H. J. Moir, and I. A. Malcolm. 2001. Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: a preliminary assessment. The Sci9ence of the Total Environment 265: 295-307.

Suttle, K. B., M. E. Power, J. M. Levine, and C. McNeely. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecological Applications 14(4): 969-974.

Townsend, C. R., M. R. Scarsbrook, and S. Dolodec. 1997. Quantifying disturbance in streams: alternative measures of disturbance in relation to macroinvertebrate species traits and species richness. Journal of the North American Benthological Society 16: 531-544.

Upper Deschutes Watershed Council. 2009. Whychus Creek restoration monitoring plan technical report. Upper Deschutes Watershed Council, Bend, OR. 44 pp.

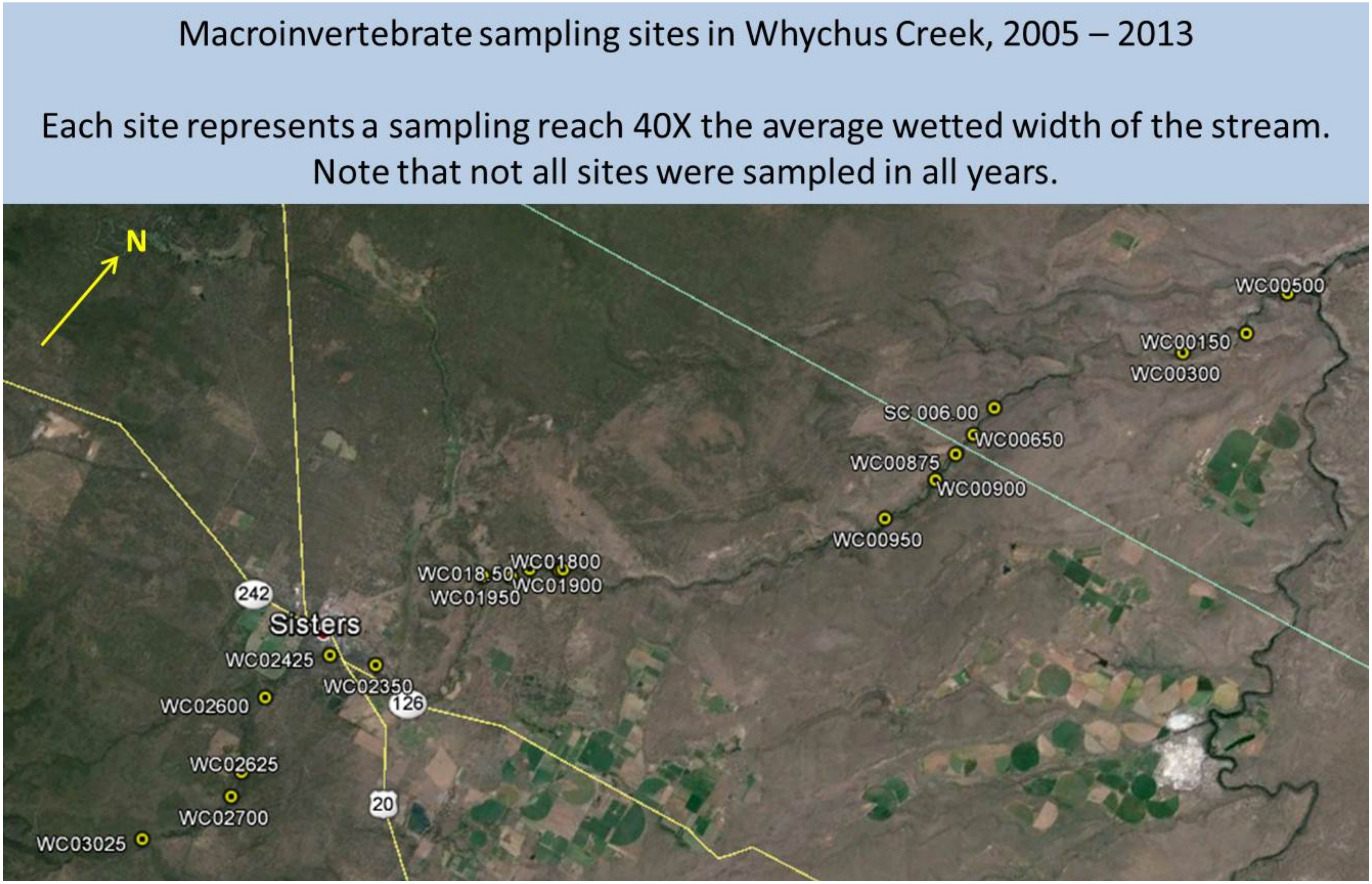
Vannote R. R., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

Ward J. W. and J. A. Stanford. 1983. Intermediate disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems. *In*: Dynamics of lotic ecosystems. Ann Arbor Science, Ann Arbor, MI. pp. 347-356.

Waters, T. F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society. Bethesda, Maryland, USA.

Wright J. F., D. W. Sutcliffe, and M. T. Furse. 200. Assessing the biological quality of fresh waters: RIVPACS and other techniques. Freshwater Biological Association. Cumbria, U. K. 373 pp.

APPENDIX A Map of sampling sites along Whychus Creek



APPENDIX B Macroinvertebrate monitoring field datasheet

Site ID ____ Date: August 17, 2013

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 ft² 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

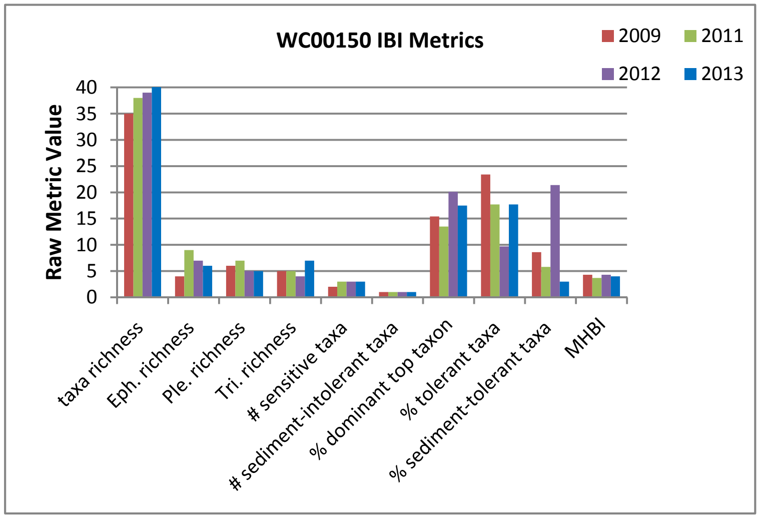
Please record units used (*check measuring tape—one side is metric, the other English*)

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

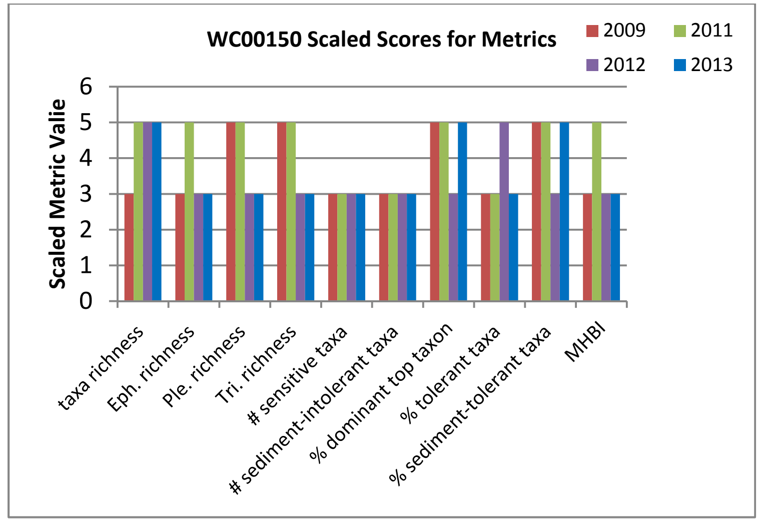
Additional notes or observations (including other wildlife noted):

APPENDIX C Site-specific changes in IBI metrics across years. Graph A is the raw value of the metric and Graph B is the scaled metric score.

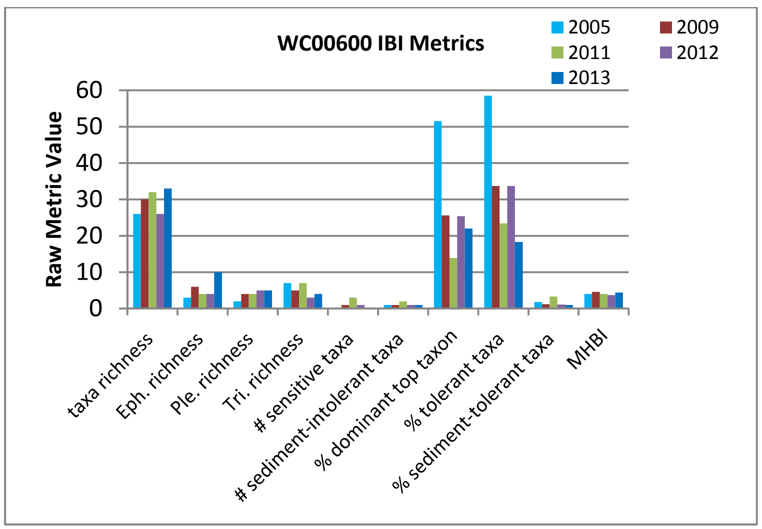
A.



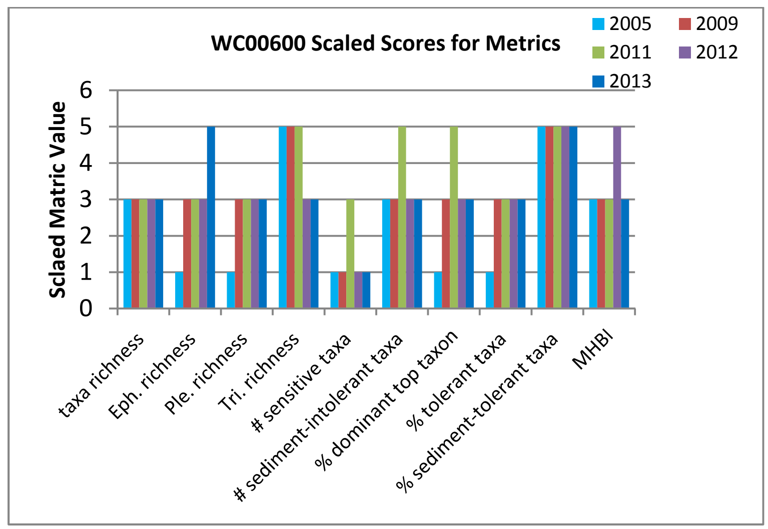
B.



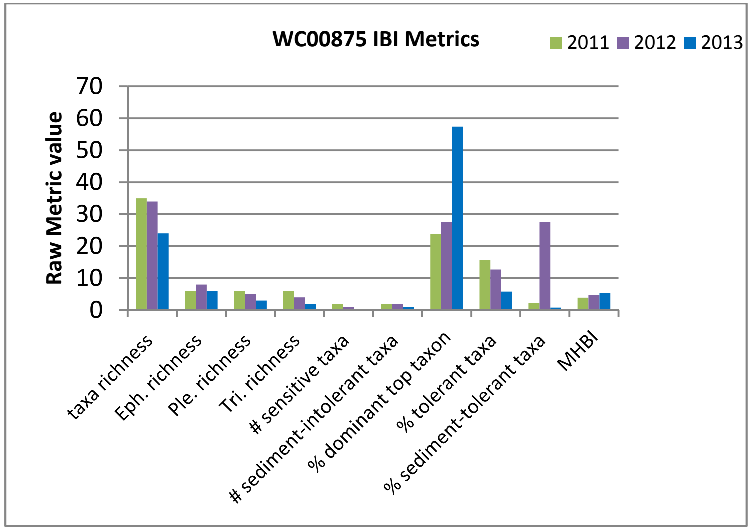
A.



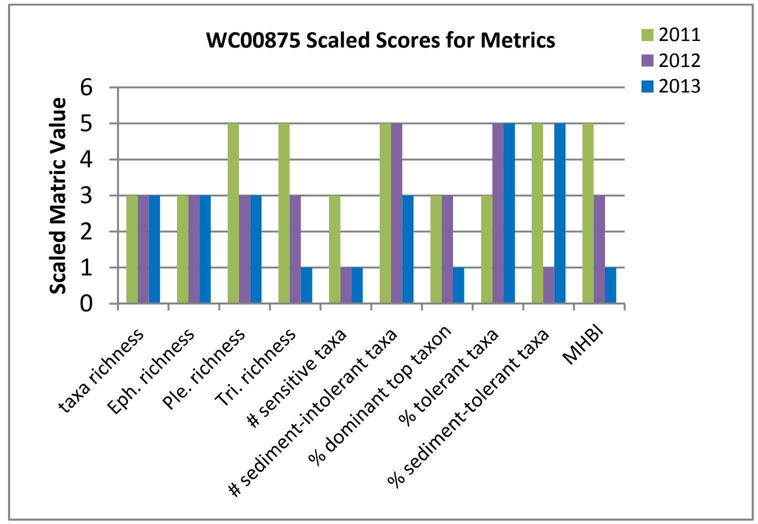
B.



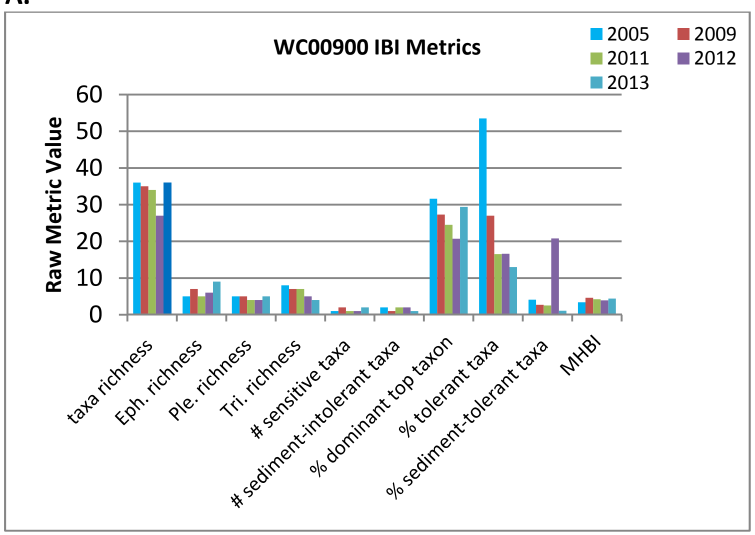
A


WC00875 IBI Metrics
2011
2012
2013
Raw Metric value
70
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

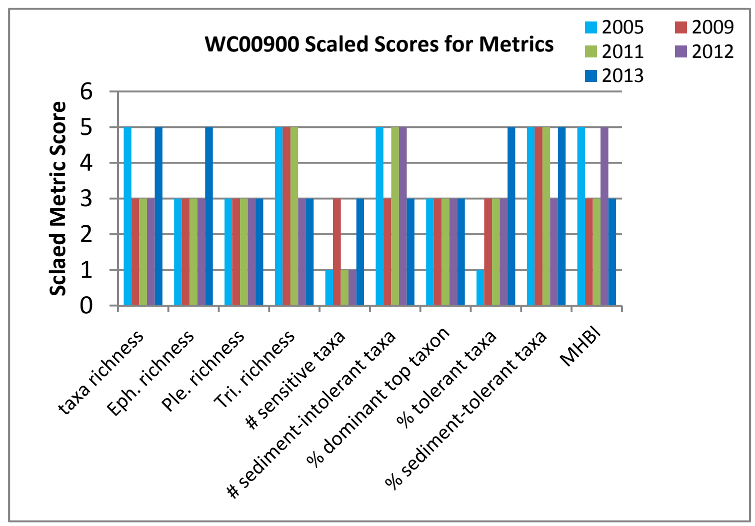
B.


WC00875 Scaled Scores for Metrics
2011
2012
2013
Scaled Matric Value
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

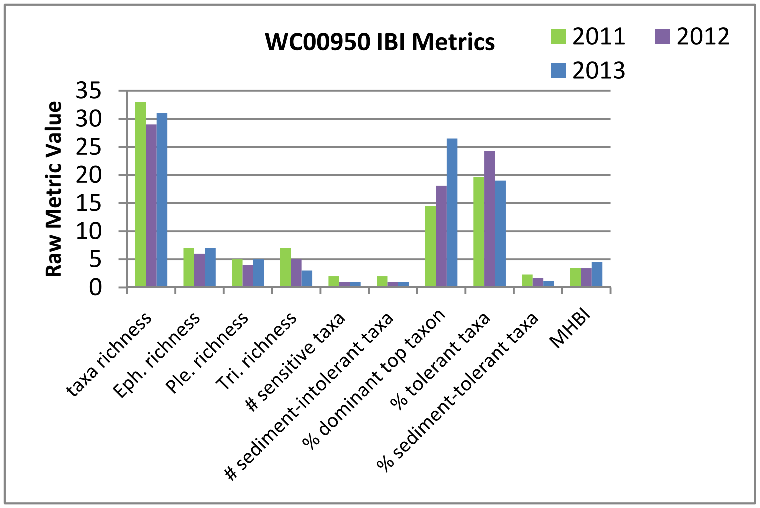
A.


WC00900 IBI Metrics
2005
2009
2011
2012
2013
Raw Metric Value
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

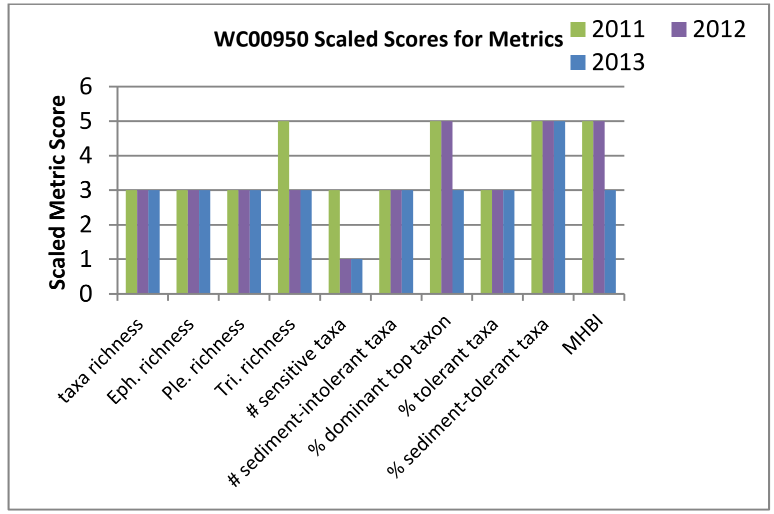
B.


WC00900 Scaled Scores for Metrics
2005
2009
2011
2012
2013
Sclaed Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

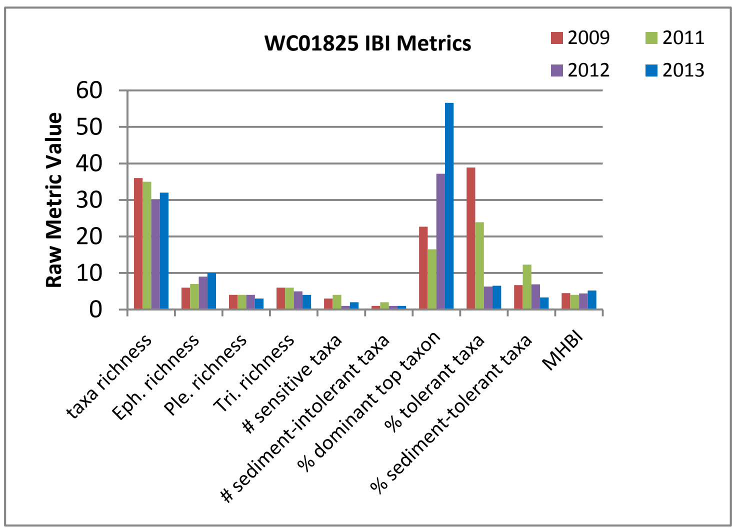
A.


WC00950 IBI Metrics
2011
2012
2013
Raw Metric Value
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

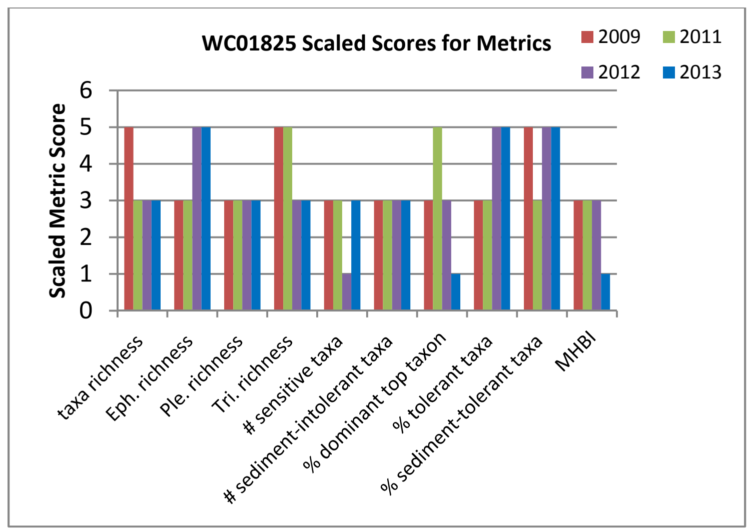
B.


WC00950 Scaled Scores for Metrics
2011
2012
2013
Scaled Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

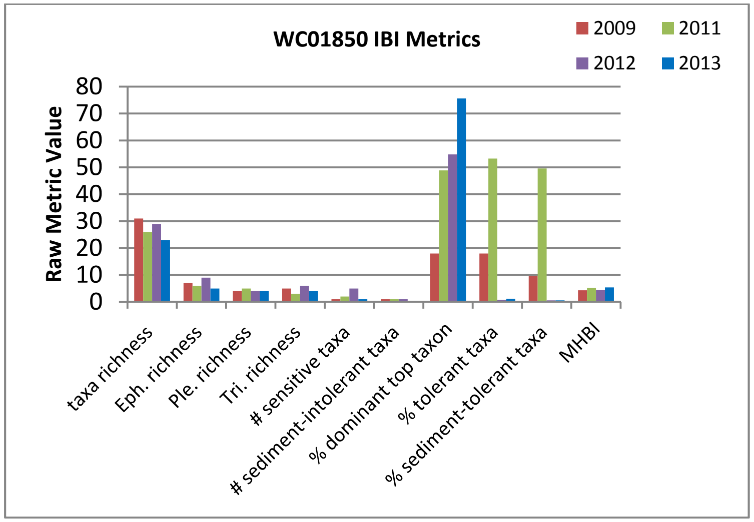
A.


WC01825 IBI Metrics
2009
2011
2012
2013
Raw Metric Value
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

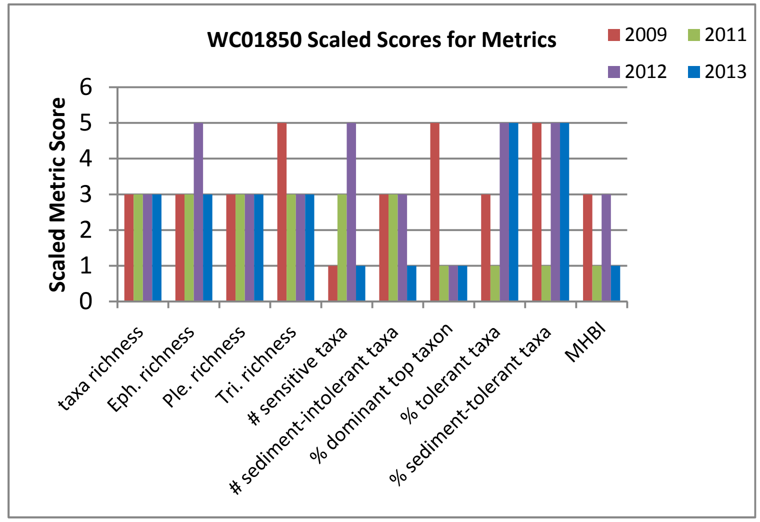
B.


WC01825 Scaled Scores for Metrics
2009
2011
2012
2013
Scaled Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

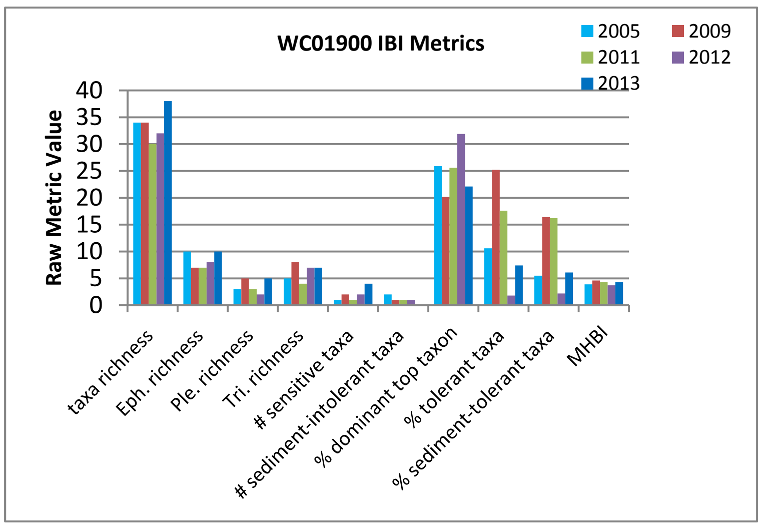
A.


WC01850 IBI Metrics
2009
2011
2012
2013
Raw Metric Value
0
10
20
30
40
50
60
70
80
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

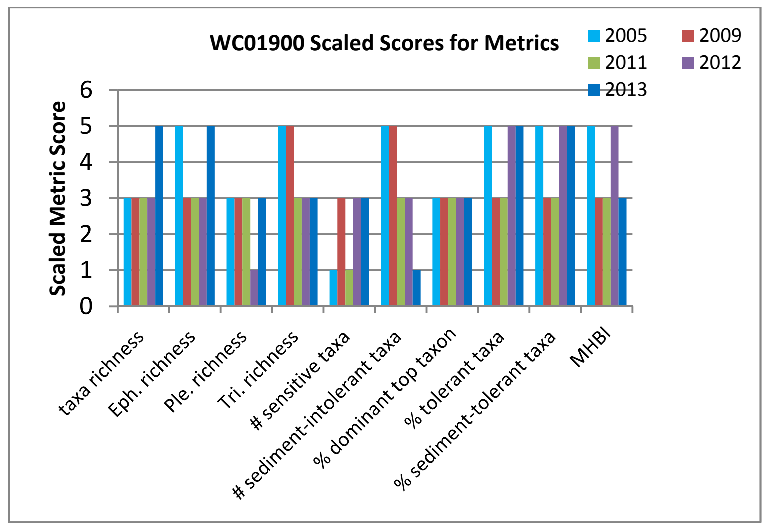
B.


WC01850 Scaled Scores for Metrics
2009
2011
2012
2013
Scaled Metric Score
0
1
2
3
4
5
6
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

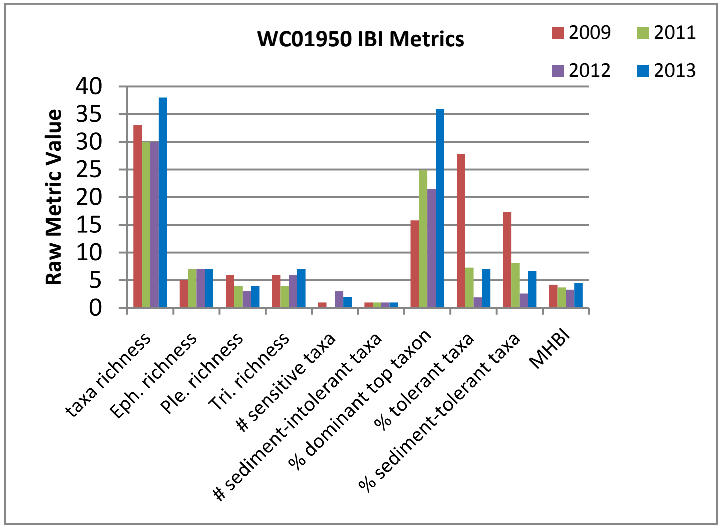
A.


WC01900 IBI Metrics
2005
2009
2011
2012
2013
Raw Metric Value
0
5
10
15
20
25
30
35
40
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

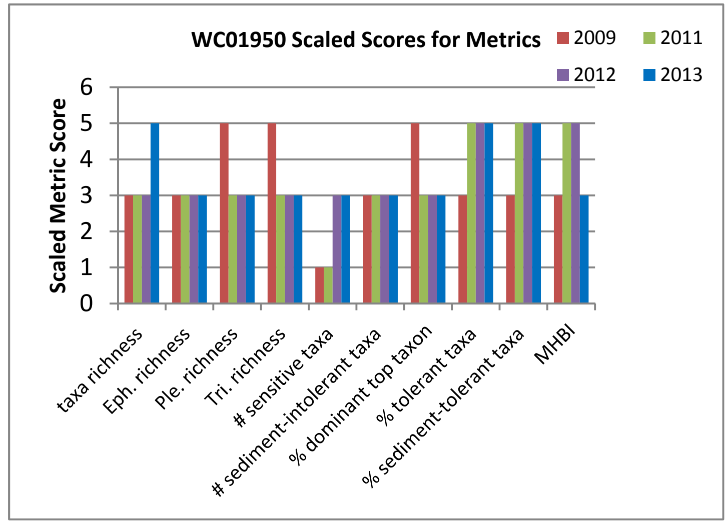
B.


WC01900 Scaled Scores for Metrics
2005
2009
2011
2012
2013
Scaled Metric Score
0
1
2
3
4
5
6
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

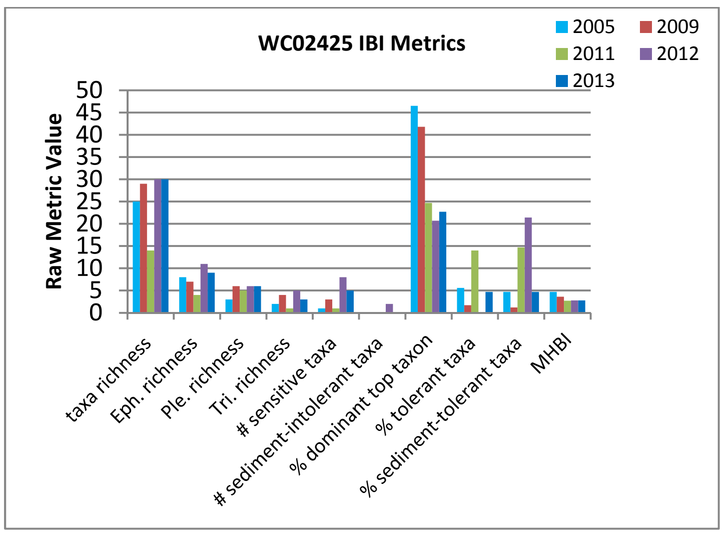
A.


WC01950 IBI Metrics
2009
2011
2012
2013
Raw Metric Value
40
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

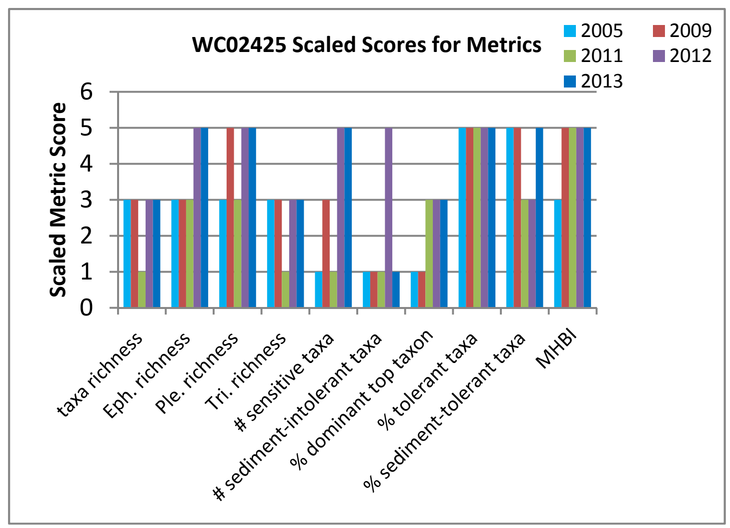
B.


WC01950 Scaled Scores for Metrics
2009
2011
2012
2013
Scaled Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

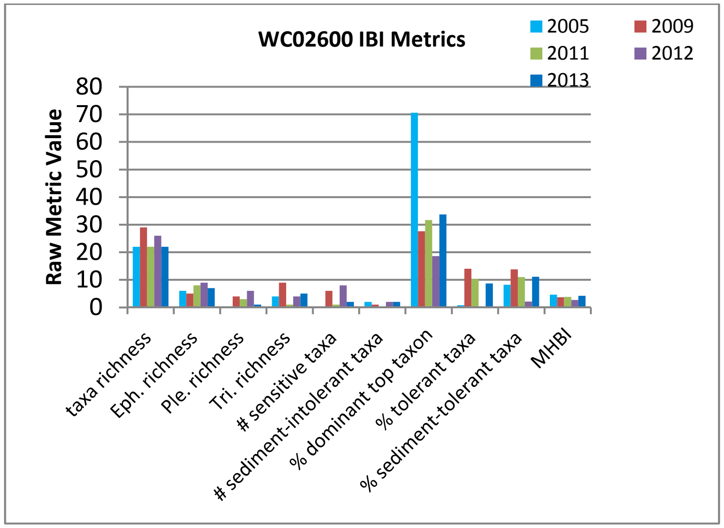
A.


WC02425 IBI Metrics
2005
2009
2011
2012
2013
Raw Metric Value
50
45
40
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

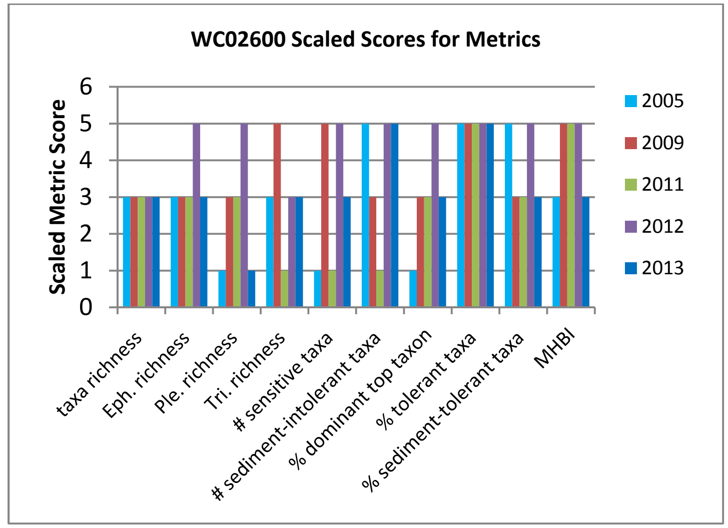
B.


WC02425 Scaled Scores for Metrics
2005
2009
2011
2012
2013
Scaled Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

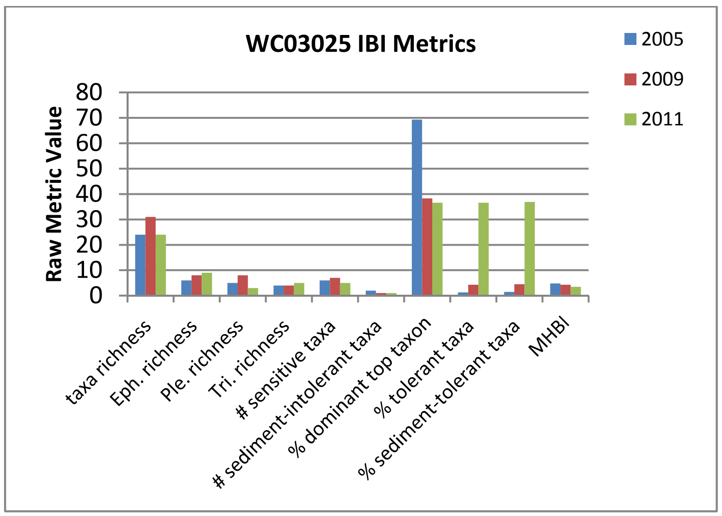
A.


WC02600 IBI Metrics
2005
2009
2011
2012
2013
Raw Metric Value
80
70
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

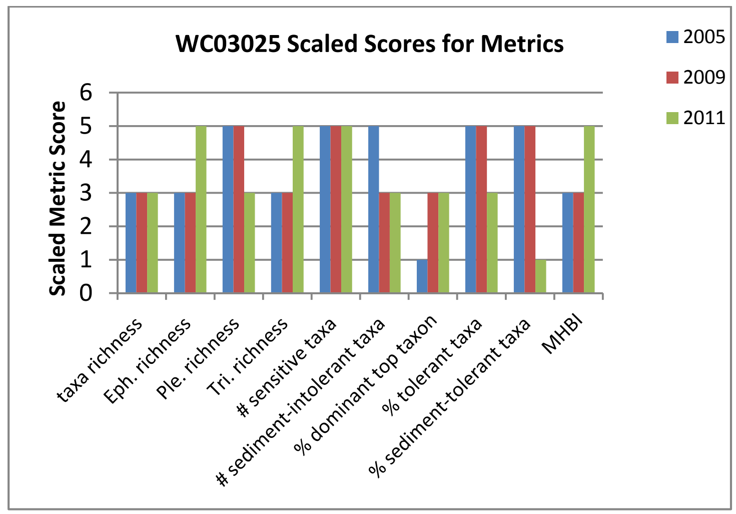
B.


WC02600 Scaled Scores for Metrics
2005
2009
2011
2012
2013
Scaled Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

A.


WC03025 IBI Metrics
2005
2009
2011
Raw Metric Value
80
70
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

B.


WC03025 Scaled Scores for Metrics
2005
2009
2011
Scaled Metric Score
6
5
4
3
2
1
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

APPENDIX D Macroinvertebrate Taxa List for Whychus Creek, 2005 - 2013

Phylum or subphylum	Class or Subclass	Order	Family	Genus	Species	2005	2009	2011	2012	2013
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	Hydroporinae/ 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	Dasyhelea				√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Forcipomyia				√		
Arthropoda	Insecta	Diptera	Blephariceridae	Bibiocephala					√	
Arthropoda	Insecta	Diptera	Blephariceridae	Blephericera		√	√	√		√
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	Baetis		√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis	tricaudatus		√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Diphetor	hageni	√	√		√	√
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella	turbida		√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ameletidae	Ameletus		√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Attenella		√	√	√	√	√
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	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	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		√				√
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	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

A suite of restoration actions on Whychus Creek aims to restore the stream habitat, flows, fish passage, and water quality necessary to support self-sustaining populations of reintroduced Chinook salmon (*Oncorhyncus tshawytscha*), steelhead trout (*Oncorhyncus mykiss*), native resident redband trout, and bull trout. Steelhead were reintroduced to Whychus Creek beginning in 2007, with Chinook first reintroduced in 2009, and hundreds of thousands of fry and thousands of smolts continue to be released annually. Restoration partners conduct native fish monitoring annually in Whychus Creek to quantify *O. mykiss* density and census *O. mykiss* redds. Portland General Electric (PGE) also traps smolts outmigrating from the tributary arms of Lake Billy Chinook, to generate smolt production estimates, and uses radio telemetry to track migration of returning adults upstream of Round Butte dam. The Whychus Creek Monitoring Plan identified fish populations as a biological indicator of restoration effectiveness. However, due to the ongoing and recently completed status of multiple restoration projects, continued annual releases of steelhead and Chinook, and the inability to differentiate between juvenile redband and steelhead pending genetic analysis, available data are inadequate to evaluate fish response to restoration. While recognizing this limitation, the Upper Deschutes Watershed Council continues to summarize PGE's and other available fish monitoring data for Whychus Creek annually to track the status and trends of fish populations. *O. mykiss* have accounted for the majority of fish caught in every year sampled since 2007. Although still relatively low, juvenile O. mykiss densities in 2013 approached or were higher than densities in any year since 2007. Juvenile spring chinook densities also remained low, but the total number of chinook caught in 2013 was higher than in previous years. Redd densities averaging 4.3 redds/km were down by almost half from the 2011 estimate of 8.3 redds/km. Smolt trapping has proven largely ineffective and captures have been insufficient to generate a smolt production estimate, however, captures of outmigrating smolts of both species between March and May provide preliminary information about smolt outmigration timing. Three adult Chinook and one adult steelhead ascended Whychus Creek in 2013; adult migration information, although limited, is beginning to identify adult migration timing in the Upper Deschutes and in Whychus Creek. Improved information about Chinook, steelhead, and redband life histories in Whychus will help restoration partners develop 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 *O. mykiss* and Chinook salmon populations and spawning activity within the study reaches, including population size, size-frequency distributions, redd counts, 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, enumerates 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

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. This 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 existing 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 and redband without conducting expensive genetic analyses. Short term changes to habitat following restoration frequently 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 will become a useful indicator of restoration effectiveness over the long term, as restored communities achieve full ecological function. PGE will conduct genetic analysis to determine relative proportions of juvenile redband and steelhead five and ten years after returning fish are first passed upstream of the hydroelectric project (2017 and 2022). These data will provide some insight into population dynamics and interactions between the two life histories. 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 re-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 2014 Fisheries Monitoring reports, including juvenile Chinook and O. mykiss rearing densities (Hill & Quesada 2014a); O. mykiss redd densities (Quesada *et al* 2014); Chinook and steelhead smolt outmigration (Hill & Quesada 2014b); and adult Chinook and steelhead returns and migration (Hill *et al* 2014). We compare 2013 data to 2007-2012 results; 2006 native fish monitoring data were collected using different methods and are not comparable to 2007-2013 data, and are therefore not considered in 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*). 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 native redband trout (*Oncorynchus mykiss*), 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) but none have been captured or observed since 2005.

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 (*Onchorhyncus 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 the rearing of sockeye juveniles. 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 migrate 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.

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. It is uncertain to what extent both life history forms will again coexist in Whychus Creek as steelhead runs are reestablished. The habitats of juvenile redband and steelhead are similar, and there will likely be some level of interaction between the two life history forms, including competition for resources and perhaps spawning interaction. 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) suggest that Whychus Creek will produce primarily anadromous, not resident, O. mykiss, based on stream flows and temperature.

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).

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. Reach 1 is located downstream from Alder Springs at river mile (rm) 1.5/river kilometer (rkm) 2.5. Reach 2 is downstream from USFS Road 6360 at rm 6 (rkm 9). Reach 3, at Camp Polk (rm 19/rkm 25.5) was sampled from 2006 through 2011 but was not sampled in 2012 following diversion of the stream from the straightened channel, where prior sampling had occurred, into the restored meadow channel. Reach 4 is downstream from Hwy 20 in Sisters at rm 23.5 (rkm 34.5). Reach 5 is located on Wolftree property at rm 17.5 (rkm 25). PGE sampled Reaches 1, 2, and 4 in 2013: PGE discontinued sampling in Reach 3 following diversion of Whychus Creek into the restored meadow channel at Camp Polk, and the crew was unable to sample in Reach 5 in 2013 due to sustained high and turbid flows.

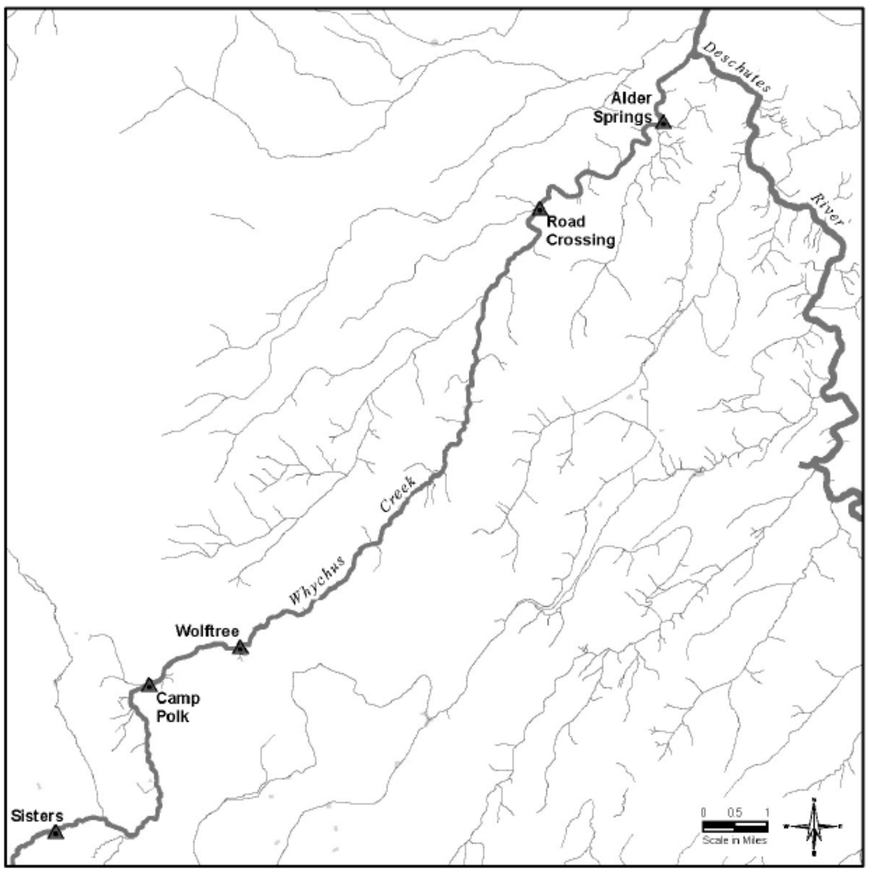

Figure 1.
Study reaches on Whychus Creek for fish population estimates. Alder Springs, Road 6360 Crossing and Sisters reaches (reaches 1, 2 and 4) have been surveyed annually since 2006; Camp Polk (Reach 3) was surveyed from 2006-2011; and Wolftree (Reach 5) was surveyed annually from 2009 through 2012. Reach 5 was not surveyed in 2013 due to high and turbid flows. Reproduced with permission from Quesada *et al* 2014**.**

PGE conducted fish population sampling during the low flow period, from September 10-19, 2013. Study reach lengths ranged from 110-223 m, determined by the location of habitat characteristics allowing the secure placement of block nets. Electrofishing habitat surveys, which highlight habitat characteristics shown to affect electrofishing efficiency (e.g. large woody debris, undercut banks), were conducted at all sampled reaches following methods used by the ODFW Research Lab and using ODFW datasheets. 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.

As in previous years, high flows in 2013 prevented effective use of block nets at some sites, and in several reaches sampling was conducted without 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 where block nets proved ineffective. 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. Where any marked fish were recaptured in the stream section below or above the reach, all fish captured in these sections were added to the Peterson 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 Quesada *et al* (2014). All fish captured were recorded by species. *O. mykiss* > 60 mm were anaesthetized, measured and marked. Chinook salmon parr were also marked where numerous enough to generate mark-recapture estimates. 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 fry releases.

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.

ODFW and USFS conducted fish and habitat surveys in one side-channel reach of Whychus Creek upstream of Sisters in 2013. These data will be analyzed with additional fish and habitat data collected along Whychus Creek in 2014, and will be included in the 2014 Native Fish Monitoring report.

O. mykiss and spring Chinook smolt production

To estimate numbers of steelhead, Chinook and redband juveniles outmigrating from Whychus Creek and other tributaries and 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. 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 attempts to trap out-migrating smolts in Whychus were abandoned. Instead, in 2011 a screwtrap was located on the Upper Deschutes River Arm of Lake Billy Chinook. In 2012, the screwtrap was replaced with a Merwin trap in the same location (Figure 2). The Merwin trap was also deployed in 2013 and operated seven days a week throughout the migratory season, from February 28th until May 30th, and checked daily. Captured fish were recorded by species. Steelhead and Chinook smolts were measured and checked for passive integrated transponder (PIT) tags. Smolts greater than 60 mm and 2.0 g without PIT tags were tagged.

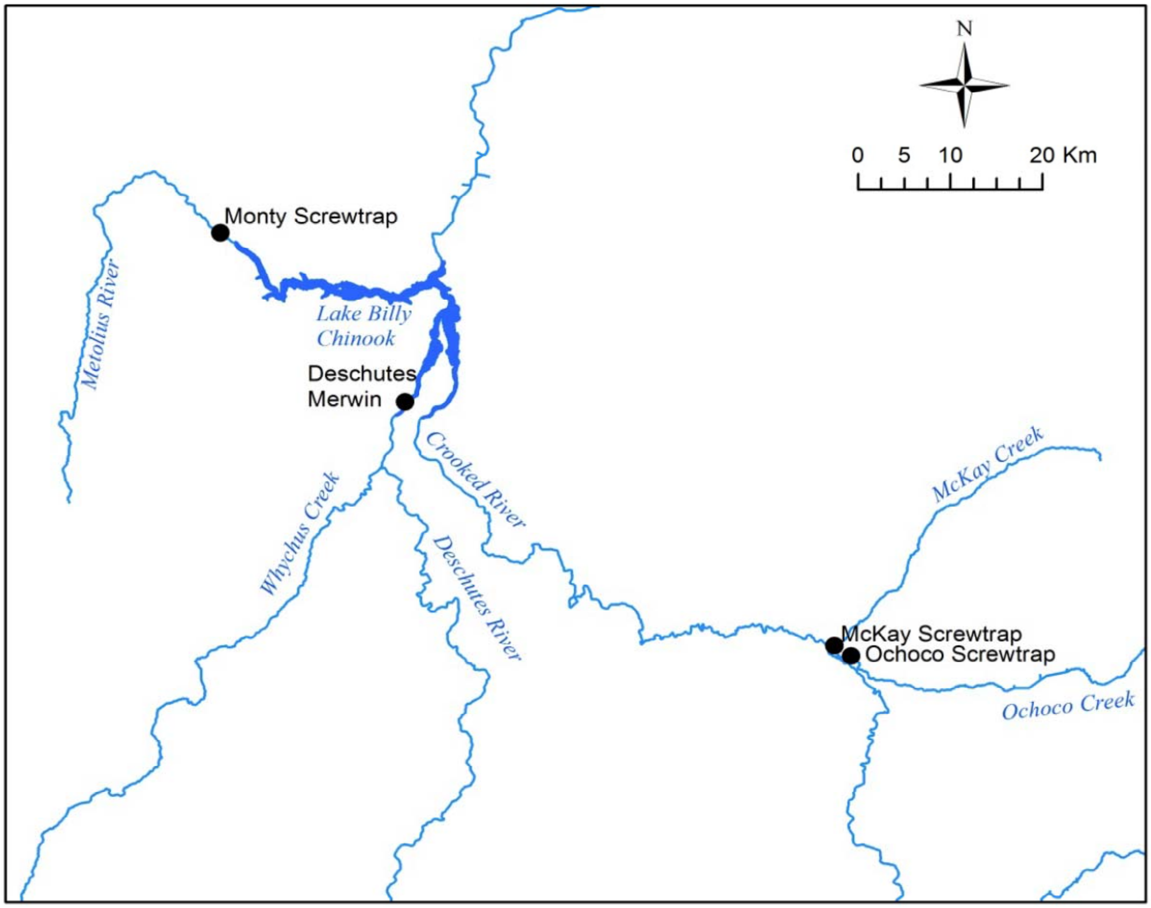

Figure 2.
2012 trap locations. Reproduced with permission from Hill and Quesada 2013.

O. mykiss redd surveys

Four locations on Whychus Creek were identified in 2006 as index sites for *O. mykiss* redd surveys (Figure 3). The four sites were subdivided into ten individual reaches to help identify the distribution of redds. PGE and the Forest Service surveyed the four sites every two weeks from March through July in 2006 – 2009. However, because spawning site selection may not be consistent or predictable between years, the reliability of index reaches to determine trends in spawning abundance has been called into question (Isaak and Thurow 2006). Additionally, changes in habitat on Whychus Creek resulting from channel reconstruction projects affecting four of the original ten survey reaches were anticipated to further diminish the suitability of data from index sites to establish spawning trends (Quesada and Hill 2010).

American Fisheries Society protocol recommends a spatially balanced rotating panel design that 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). This design, similar to that used by the ODFW Coastal Salmonid Inventory Project (ODFW 2007), allows estimates of redds per kilometer and spawning distribution, and reduces bias. In 2010 PGE revised redd count methods to incorporate the rotating panel design. From 2010 through 2012 PGE conducted redd counts in a total of eight reaches each year. Two designated 1-km index sites corresponding to reaches 1 (Alder Springs, rkm 2) and 8 (immediately upstream of Camp Polk, rkm 27) were surveyed each year, and every year two additional 1-km sites were randomly selected from thirty 1-km reaches between the mouth of Whychus Creek and Sisters. In addition, four of the ten original reaches were surveyed annually to help establish a population trend and to identify the temporal and spatial *O. mykiss* spawning distribution.

In 2013 PGE conducted redd counts in six reaches: in the two index sites, one at Alder Springs (rkm 2) and one immediately upstream of Camp Polk (rkm 27); in one randomly selected reach at rkm 21; and at three additional sites: the restored meadow channel at Camp Polk (rkm 25; reach 5), Lewis Woodpecker Creek (reach 3), and Alder Springs Cree (reach 2)(Figure 2). The Camp Polk reach was surveyed in place of a second randomly chosen reach to allow evaluation of fish use of the new channel. 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.

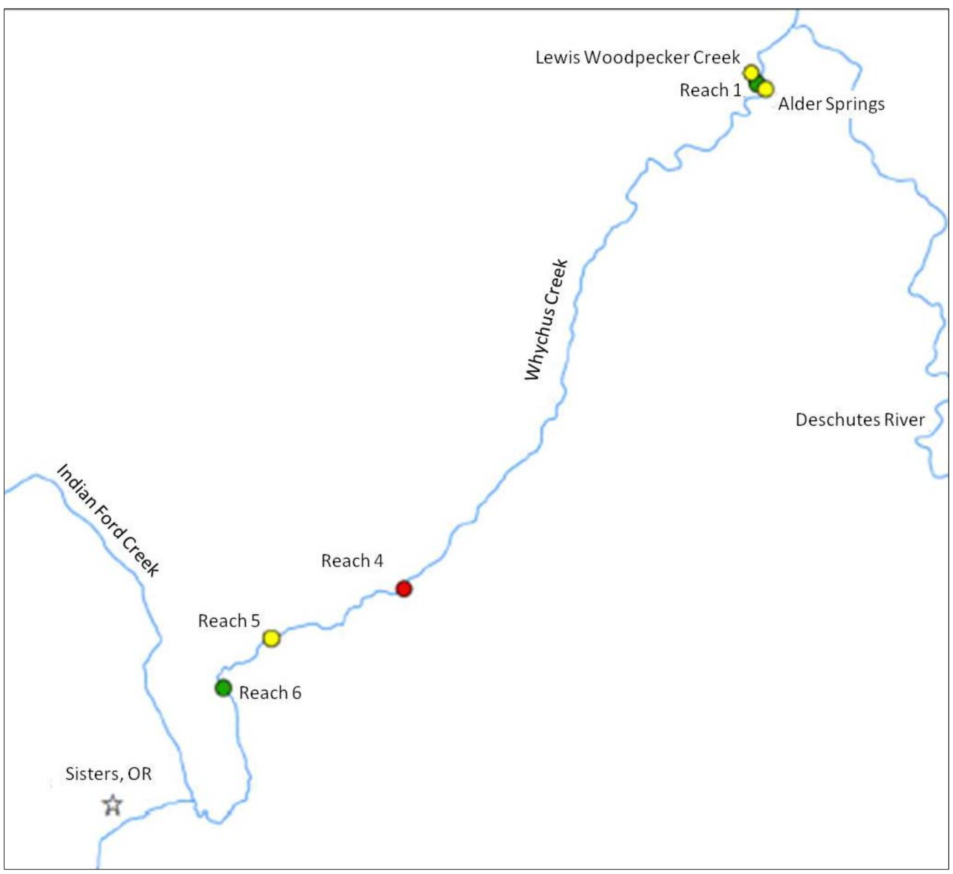

Figure 3.
Redband redds were counted in six reaches in 2013: two designated index reaches (green circles: reach 1, rkm 2; reach 6, rkm 27); one randomly selected reach (red circle: reach 4, rkm 21); and in three original reaches (yellow circles: reach 2, Alder Springs Creek; reach 3, Lewis Woodpecker Creek; and reach 5, Camp Polk). Reaches 2 and 3 were surveyed from 2006-2012; reach 5 was surveyed from 2006 through 2013. Reproduced with permission from Quesada *et al* 2014.

Adult returns and migration

Numbers of adult steelhead and Chinook returning to the Pelton trap were recorded by date. In 2013, 50% of returning adult steelhead and all returning 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 passed upstream of Round Butte dam. Thirty-four adult steelhead and 23 adult chinook were tagged with Juvenile Combined Acoustic Radio Telemetry 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 additional stations on all three rivers. 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 2013 the majority of fish captured in Whychus Creek were *O. mykiss* including both resident redband and released steelhead. One bull trout approximately 355 mm in length was captured in the Alder Springs reach (Reach 1). Other species captured included Chinook salmon, brown trout, sculpin, and longnose dace.

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 seven years of sampling (Table 2, Figure 4). Estimated densities for any year or reach have rarely exceeded 40 *O. mykiss*/100m^2. However, 2013 densities approached or were higher than in any year since 2007. Size distribution of *O. mykiss* from 2007-2013, 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.

Table 2. O. mykiss density estimates from 2007-2013. 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
1 (Alder Springs)	48 (± 28)	24 (± 24)	12 (± 4)	11 (± 4)	24 (± 5)	7 (± 2)	27 (± 2)
2 (Road 6360)	25 (± 10)	9 (± 3)	24 (± 9)	13 (± 4)	15 (± 3)	15 (± 3)	31 (± 3)
3 (Camp Polk)	60 (± 13)	52 (± 21)	57 (± 15)	27 (± 9)	16 (± 3)	-	-
4 (Sisters)	20 (± 10)	5 (± 2)	23 (± 14)	18 (± 6)	10 (± 5)	28 (± 8)	27 (± 8)
5 (Wolftree)	-	-	21 (±7)	106 (± 29)	42 (± 9)	32 (± 9)	-
USFS site at TSID	-	-	2.4 (1.5-4.0)	-	-	-	-

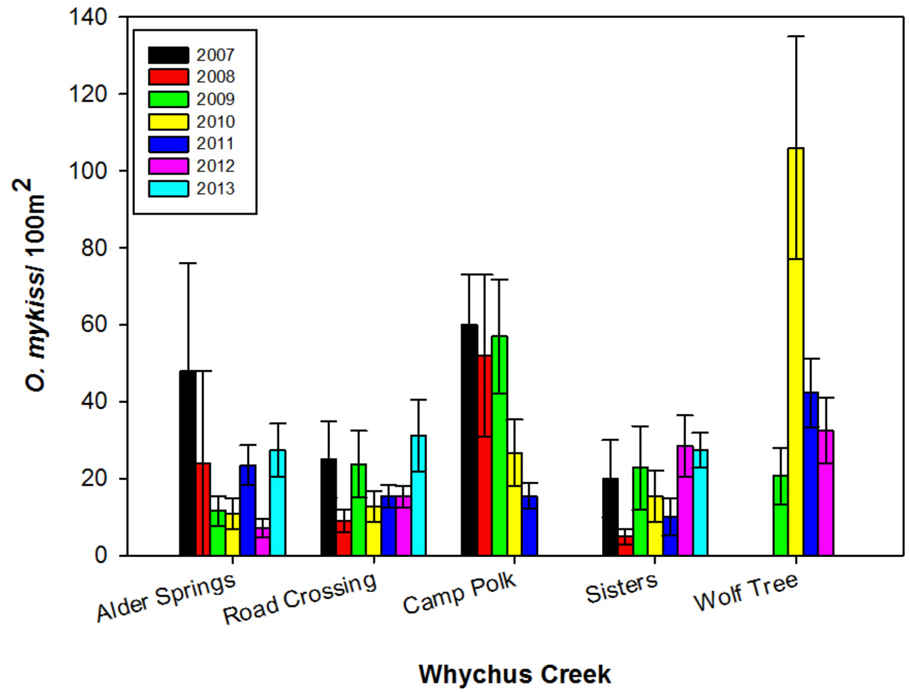

Figure 4. *O. mykiss* densities for five Whychus Creek sites from 2007 to 2013. Reproduced with permission from Quesada *et al* 2014.

Spring Chinook juvenile density & size

Juvenile spring Chinook densities in Whychus Creek remained low in 2013, but the total number caught was higher than in previous years (Table 3, 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 3. Spring Chinook densities in Whychus Creek in 2009-2013 estimated from mark-recapture and snorkel surveys.

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

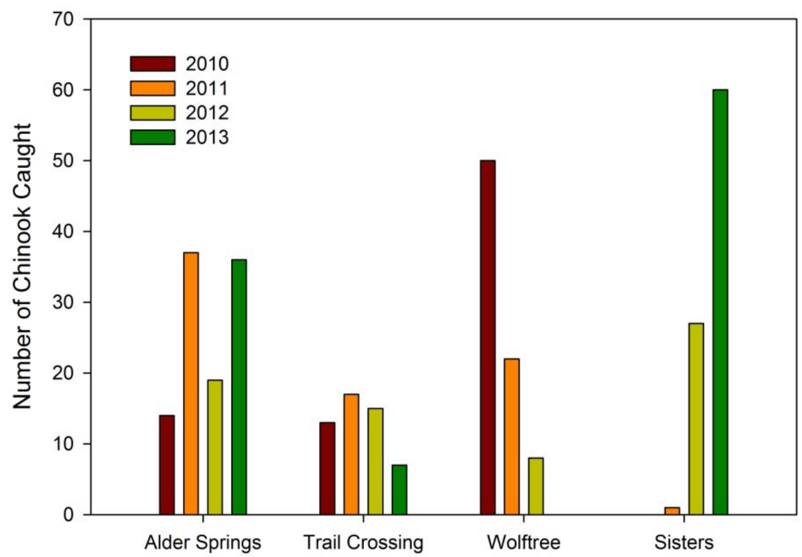

Figure 5. Number of Chinook caught during Whychus electrofishing surveys from 2010-2013. Reproduced with permission from Hill & Quesada 2014

Juvenile migration and smolt production

In 2013 as in 2012, fish captures at the Merwin trap on the Upper Deschutes River arm of Lake Billy Chinook were too low to generate a smolt production estimate for the Upper Deschutes and Whychus Creek. A total of 25 naturally reared and 35 hatchery reared steelhead smolts were caught at the Deschutes River Arm trap between March and May, primarily in April and May. Chinook smolt catch totaling 11 naturally reared and 36 hatchery reared fish peaked in mid-April, with captures from March through May.

Adult returns and migration

One hundred and thirty-three adult steelhead were captured at the Pelton Trap between September 2012 and May 2013, peaking in November 2012. Twenty-two adult Chinook salmon returned to the Pelton Trap between late April and August, 2013, peaking in late June. Thirty-four of the adult steelhead and all 22 adult chinook were JCART-tagged and released upstream of Round Butte Dam. Of these, three steelheaded (9%) ascended the Deschutes River Arm from Lake Billy Chinook; two were detected in the Deschutes near the confluence with Whychus; the third ascended Whychus to Rimrock Ranch (rkm 15). The fish was not found but its tag was recovered near a large redd with measurements larger than redband redds measured prior to releases of adult steelhead upstream of Round Butte dam. Five of the 22 Chinook (23%) ascended and remained in the Deschutes River system, with a median travel time of 2.5 days (range 1-34 days). Four spring Chinook were detected in the Deschutes River and three ascended Whychus Creek. One was located upstream of Road 6360 (rkm 9), one at rkm 4 between Alder Springs and Road 6360, and one at Rimrock Ranch (rkm 15).

O. mykiss redd surveys

Surveyors identified a total of 25 redds in Whychus Creek in 2013 (Figure 6). As in previous years, the majority of redds were located in reaches 1-3, in the vicinity of Alder Springs. Redds averaged 4.3/km in 2013, down from 8.3 redds/km in 2011. (No average density was available for 2012 due to high turbidity and consequently low visibility which reduced the effectiveness of redd counts.) One redd, located in the vicinity of a recovered radio tag shed by a returning adult steelhead, was substantially larger in pot length, width, and tail spill width than redband trout redds measured in 2013. Spawning in Whychus was recorded from March through June and peaked in May.

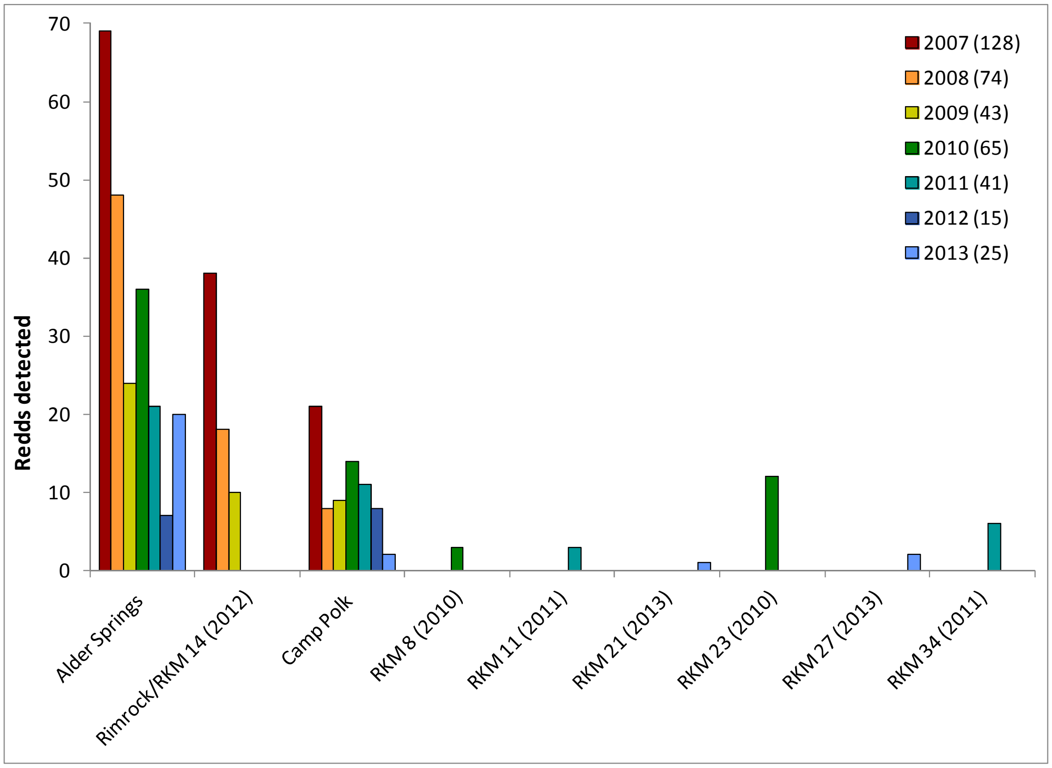

Figure 6. Redband redds detected by site and year in Whychus Creek. Totals for each year are shown in the legend. RKM 38 was sampled in 2012 but is not shown because no redds were found at that location.

Discussion

Population estimates

Despite annual releases averaging 271,870 fry and 4,230 smolts, far higher than in McKay and Ochoco Creeks, juvenile *O. myki*s density remains low in Whychus, and far lower than (approximately half) the densities estimated for the two Crooked River tributaries (Quesada *et al* 2014). Densities in the three reaches sampled in Whychus were approximately equivalent; and, while overall juvenile *O. mykiss* densities in Whychus remained low in 2013, densities at the two downstream sites (Alder Springs and Road 6360) increased by approximately half to two-thirds over 2012 numbers. The estimated density in the Sisters reach was slightly lower than in 2012. With the exception of the Sisters site in 2012, densities at all three Whychus sites were higher in 2013 than in any year since 2007, despite the number of fry released in 2013 being similar to previous years. Chinook densities also remain lower in Whychus Creek, than in all other rivers and tributaries sampled (Metolius, Lake Creek, Ochoco Creek).

Poor spawning and emergence conditions along 32% of river miles, and stream temperatures that are too high to support spawning (Mork 2013, Mork 2014) could be limiting spawning and emergence of resident native redband, resulting in a small contribution of the resident population to O. mykiss juvenile density. But, given the high numbers of steelhead fry released in Whychus Creek annually, low resident production is an unlikely explanation for the low and variable juvenile *O. mykiss* numbers observed. Habitat quality for rearing and migration, rated fair or good along the extent of the creek, also does not explain the low O. mykiss densities in Whychus.

Little explanation exists for the continued low density of juvenile O. mykiss and spring Chinook salmon in Whychus Creek. Stream conditions, specifically elevated temperatures that exceed biological requirements for rearing fish, may simply be inadequate to support survival. Macroinvertebrate data from 2005 to 2013 (Mazzacano 2013) also indicate some level of impairment at most sites, suggesting compromised stream conditions that may support lower numbers of juvenile O. mykiss and spring Chinook than otherwise anticipated. 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. Despite the continued low estimates for *O. mykiss* 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 and smolts.

The inability to differentiate between juvenile redband and steelhead continues to limit conclusions about the relative abundance of the two life histories and interactions between the two populations. In 2013 as well as in 2010, PGE collected genetic samples from a subset of *O. mykiss* captured during electrofishing. These samples will be used in the future to estimate the proportion of resident redband and reintroduced steelhead comprising the population, determine the rearing success of hatchery outplants, and quantify the rate of residualization of reintroduced stock. PGE will conduct 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. The ability to differentiate between juvenile steelhead and redband will allow researchers to better understand the status and trends of resident redband and reintroduced steelhead populations, potential effects of competition between the two life histories, and spawning interactions.

Life histories 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 to support each life history. Redd counts conducted between 2007 and 2013 show redband spawning peaking in May, with the greatest concentration of redds consistently located in the Alder Springs reaches and at Rimrock Ranch, recommending that flows are managed to maintain temperatures below 13°C in these reaches through the end of May and into June as possible. Data from 2009 through 2013 show steelhead smolt outmigration occurring in April and May, with Chinook outmigration having peaked in both March and April; although the temperature criteria for migration is 18°C, redband spawning during the same months requires the lower 12.8°C temperature. Because steelhead characteristically outmigrate at 2-3 years of age, once returning steelhead begin spawning in Whychus Creek and following emergence of the first cohort, steelhead rearing will occur year-round.

Conclusions

Although fish population data from 2007 to 2013 reveal no clear O. mykiss population trends in Whychus Creek, the decline in *O. mykiss* density indicated by 2007-2011 data has reversed in 2012 and 2013, with a 2013 average density estimate that is the third-highest of the seven year dataset. And, although lower than in any year but 2012, the number of redband redds also increased over the 2012 number. Average spring Chinook density estimates remained low in 2013.

Native fish monitoring results reported for Whychus Creek demonstrate the frequent challenges associated with monitoring fish populations. Seemingly low numbers for fish population metrics may reflect methodological challenges encountered sampling in Whychus Creek, or they may reflect the true condition of these populations, or some combination thereof. Low fish densities in Whychus Creek could be a product of the influence of a number of varied environmental, climatic, and biological factors, possibly including historic habitat alteration, the magnitude and timing of high flow events, stream productivity, population dynamics associated with steelhead and salmon releases, and short-term impacts of stream channel restoration projects.

Ongoing refinements of PGE native fish monitoring protocols will continue to improve the utility of the resulting data to describe fish populations in Whychus Creek. Increased accuracy of population estimates will improve our understanding of Chinook salmon and *O. mykiss* abundance across years; the revised redd sampling design will provide a basis to establish spawning distribution and abundance of redband and steelhead. As researchers continue to gain experience with trapping locations and trap operations, more and higher quality data will become available to generate an estimate of Whychus smolt production. Genetic analysis will eventually allow researchers to describe juvenile resident redband and reintroduced steelhead abundance and trends, and interactions between the two populations.

Restoration partners initially expected that biological indicators would provide an effective means for evaluating trends in watershed restoration. In the short term, the data available on fish populations and especially *O. mykiss* 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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References

Ackerman NK, Justice C, Cramer S. 2007. Juvenile steelhead carrying capacity of the Upper Deschutes Basin. Cramer Fish Sciences, Sandy, Oregon.

Behnke RJ. 2002. Trout and salmon of North America. Free Press, Simon and Shuster Inc. New York, New York. 359 p.

Christie MR, Marine ML, Blouin MS. 2011. Who are the missing parents? Grandparentage analysis identifies multiple sources of gene flow into a wild population. Molecular Ecology 20:6.

Cramer SP, Beamesderfer RCP. 2006. Population dynamics, habitat capacity, and a life history simulation model for steelhead in the Deschutes River, Oregon. Cramer Fish Sciences, Sandy, Oregon.

Dachtler N. 2007. “Existing condition stream survey and fish population report.” Appendix E in Whychus Creek Restoration Project at Camp Polk Meadow Preserve: a collaborative project of the Upper Deschutes Watershed Council, Deschutes Land Trust, and Deschutes National Forest. Bend, OR.

Fies T, Fortune J, Lewis B, Manion M, Marx S. 1996. Upper Deschutes River sub-basin fish management plan. Oregon Department of Fish and Wildlife, Portland, Oregon.

Gallagher SP, Hahn PKJ, Johnson DH. 2007. Redd counts. Pages 199-234 in Johnson DH, Shrier BM,

Groves K, Shields B, Gonyaw A. 1999. Lake Billy Chinook rainbow (redband) trout life history study – final report. Oregon State University, Corvallis, Oregon, and Portland General Electric Company, Portland, Oregon.

Hill M. 2009. Personal Communication. Fishery biologist, Portland General Electric Company, Round Butte Pelton Project Office, Madras, Oregon.

Hill M. 2011. Personal Communication. Message to Lauren Mork. December 9, 2011. Fishery biologist, Portland General Electric Company, Round Butte Pelton Project Office, Madras, Oregon.

Hill M, Burchell R, Bennett M, Wymore B, Quesada C. 2014. 2013 Adult migration, survival, and spawning test and verification annual report. Portland General Electric Company, Portland, Oregon.

Hill M, Quesada C. 2014a. 2013 Annual Test and Verification Report: Salmonid Rearing, Juvenile Density, Habitat. Portland General Electric Company, Portland, Oregon.

Hill M, Quesada C. 2014b. 2013 Juvenile Migration Test and Verification Study Annual Report. Portland General Electric Company, Portland, Oregon.

Isaak DJ, Thurow RF. 2006. Network-scale spatial and temporal variation in Chinook salmon (*Oncorhynchus tsawytscha*) redd distributions: patterns inferred from spatially continuous replicate surveys. Canadian Journal of Fisheries and Aquatic Sciences. 63:285-296.

Kunkel C. 2010. “Whychus Creek baseline technical report: native fish monitoring.” Pages 120-134 *in* Golden B, Houston R, Editors. 2009 Whychus Creek monitoring report. Upper Deschutes Watershed Council, Bend, OR.

Lewis SD. 2003. *Onchorhyncus mykiss* monitoring in Squaw Creek. Portland General Electric Company. Portland, Oregon.

Mazzacano CA. 2013. Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, and 2011 – 2013. Prepared for the Upper Deschutes Watershed Council, Bend, Oregon.

Mork L. 2013. “Habitat Quality in Whychus Creek.” Pages 47-62 in Mork L, Houston R, Editors. 2011 Whychus Creek Monitoring Report. Upper Deschutes Watershed Council, Bend, Oregon. 121 p.

Mork L. 2014. Whychus Creek Water Quality Status, Temperature Trends, and Stream Flow Restoration Targets. Upper Deschutes Watershed Council, Bend, Oregon. 26 p.

Nehlsen W. 1995. Historical salmon and steelhead runs of the Upper Deschutes River Basin and their environments. Madras, Oregon. 65 p.

ODFW (Oregon Department of Fish and Wildlife). 2007. Coastal salmon spawning survey procedures manual 2007. Oregon Department of Fish and Wildlife, Corvallis, Oregon.

ODFW (Oregon Department of Fish and Wildlife) and CTWS (Confederated Tribes of Warm Springs). 2008. Reintroduction and conservation plan for anadromous fish in the Upper Deschutes River sub-basin, Oregon. 78 p.

Quesada C, M Bennett, and M Hill. 2014. Native Fish Monitoring: Biological Component 2013 Annual Report and 2014 Work Plan. Portland General Electric Company, Portland, Oregon.

Riehle MD and Lovtang J. 2000. Squaw Creek Education and Restoration Project 1998-1999. Deschutes National Forest. Sisters, OR.

Scheerer P, Gunckel S, Jacobs S. 2007. Redband trout, Warner sucker, and Goose Lake fishes distribution and abundance survey protocols. Oregon Department of Fish and Wildlife, Research and Development Section, Corvallis, Oregon.

UDWC (Upper Deschutes Watershed Council). 2009. Whychus Creek restoration monitoring plan. Technical report. Upper Deschutes Watershed Council, Bend, Oregon. 40 p.

Zimmerman CE and Reeves GH. 1999. Steelhead and rainbow trout early life history and habitat use in Deschutes River, Oregon (Final Report). Portland General Electric Company, Portland, Oregon.

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]	12.8°C												
	Egg Incubation & Emergence													
	Juvenile Rearing	18°C												
	Juvenile Outmigration													
	Adult In-migration													
Steelhead	Spawning[a]	12.8°C												
	Egg Incubation & Emergence													
	Juvenile Rearing[b]	18°C												
	Juvenile Outmigration[c]													
	Adult In-migration[d]													
Chinook	Spawning	12.8°C												
	Egg Incubation & Emergence													
	Juvenile Rearing	18°C												
	Juvenile Outmigration[c]													
	Adult In-migration[d]													

Peak activity

Activity documented

Predicted activity

[a] Source: PGE redd count data, 2007-2013

[b] Quinn 2005

[c] Hill & Quesada 2014b

[d] Hill *et al* 2014