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Technical Report

2014 Whychus Creek Monitoring Report

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Symbols and Abbreviations

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

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

Restoration Effectiveness Monitoring in Whychus Creek

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Introduction

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

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

As restoration investments and commitments increased, restoration partners 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 accomplished as efficiently as possible by using existing data. The reliance on existing data inherently limits the types of analyses and the conclusions that can be developed.

The monitoring approach selected by UDWC focuses on tracking the status and trends of key physical and biological indicators in Whychus Creek. We selected these indicators based on a conceptual model of factors limiting salmonid production in the creek (Figure 1), on the premise that ongoing restoration actions will ameliorate the limiting factors identified in the conceptual model and that selected indicators will respond to changes in these limiting factors. This approach will not test cause and effect relationships between restoration actions and changes in selected indicators. It will, however, demonstrate whether these indicators have moved closer to desired conditions. We drew indicators from seven broad categories: stream flow, water quality, habitat quality, stream connectivity, fish entrainment, macroinvertebrates, and fish populations. Each individual section of the 2014 Whychus Creek Monitoring Report assesses indicators in one of these categories.

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

Study Area

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

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

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

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

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

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

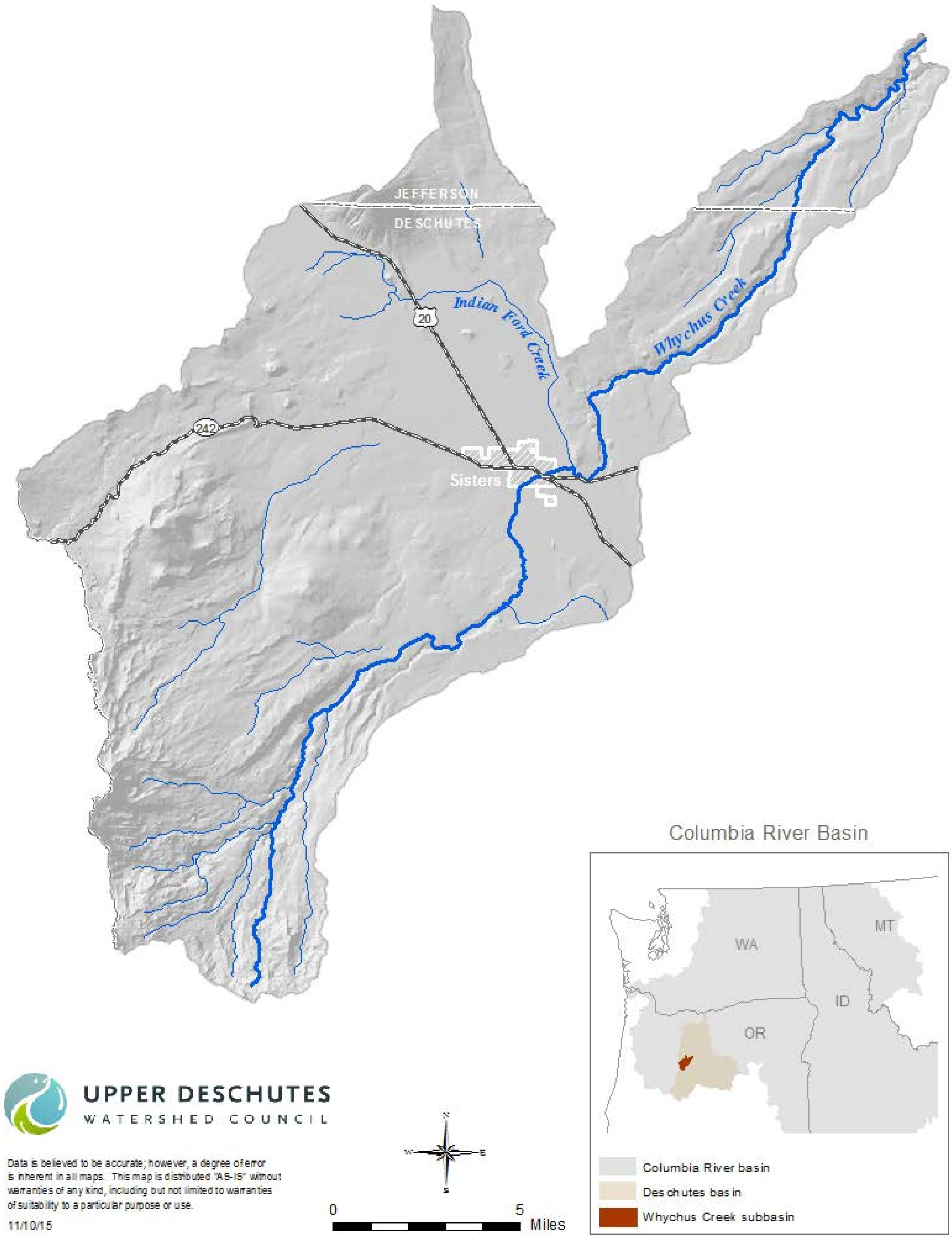

Figure 2.
Whychus Creek extends from the Cascade Range to the Deschutes River. The creek's watershed encompasses approximately 162,000 acres of Deschutes and Jefferson Counties in central Oregon.

Technical Studies

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

Stream flow and stream temperature metrics are anticipated to change in response to stream flow restoration. Golden & Wymore (2015) evaluate low flow metrics to track summer stream flow trends in Whychus Creek from 2000-2014. The 30-day moving average minimum flow represents the lowest flow conditions over the course of a year and indicates when the worst flow conditions occur, contributing to our understanding of when flow restoration is most critically needed. 30-day moving average minimum flows have steadily increased since 2001, most frequently occurring in August (36%; five years), followed by September (29%; four years) and May (14%; two years). The 2014 30-day moving average minimum flow was 22 cfs, on October 6 and 7.

Mork (2015a) reports on changes in stream temperature in relation to state standards for trout rearing and migration (18°C) and salmon and steelhead spawning (12.8°C), and discusses the relationship between stream flow and stream temperature in Whychus from 2000 - 2014. Stream temperature (seven day moving average maximum; 7DMAX) was warmer than the standard for rearing trout, exceeding 18°C at five sites representing 12 miles and for up to 59 days in 2014, the second-highest number of days in the last 7 years; but, temperatures remained below the range at which growth is severely impaired, with no 7DMAX temperature higher than 21˚C in 2014. Warm stream temperatures in excess of 12.8°C at six sites (13.5 miles) for 3-13 (6-39%) spawning season (January 1-May 15) days for which data were available may also have compromised spawning conditions. The updated stream flow-temperature relationship for the most impaired site on Whychus Creek shows 62 cfs of instream flow needed to meet the 18°C temperature standard in July.

Two reports quantify habitat improvements achieved through restoration projects. Mork (2015b) documents the status of stream connectivity along the creek resulting from removal or retrofitting of fish passage barriers as a measure of stream connectivity along the creek. As of 2014, the number of stream miles accessible from the mouth of the creek has increased from the 2009 baseline of 15.5 miles to 26.8; the number of fragmented reaches reduced from seven to two; and the number of barriers reduced from six to one. Mork (2015c) presents reductions in fish entrainment potential on the creek, using unscreened irrigation diversions and the cumulative flows diverted through them as a proxy for entrainment potential. Four diversions were screened between 2009 and 2014; flows diverted through unscreened diversions were reduced by 87%, from 192.9 to 22.5 cfs. Restoration partners plan to provide passage at the last remaining barrier and screen or decommission five of the remaining eight unscreened diversions identified in the 2009 baseline report. Although stream channel and floodplain restoration projects are expected to improve fish habitat on Whychus, no new habitat data are available to evaluate changes in habitat quality resulting from these projects (but see Mork 2012 for the most current habitat quality ratings for Whychus).

The final two reports update the status of biological conditions in the creek. Mork (2015d) summarizes findings from 2014 PGE and USFWS O. mykiss and Chinook monitoring. This report outlines the status of O. mykiss populations and Chinook reintroduction. While fish monitoring data from Whychus cannot at present provide information about how restoration is changing stream conditions, they do contribute insight into population dynamics and productivity in Whychus. Density of juvenile O. mykiss has remained consistently low in Whychus (average density in 2014: 27 fish/100m^2) with no significant increasing or decreasing trend despite annual releases of hundreds of thousands of fry; resident O. mykiss spawning has fallen dramatically since reintroduction began; and 2014 genetic analyses show that the juvenile O. mykiss present in Whychus represent predominantly (75%) Round Butte Hatchery stock, indicating reintroduced fry have residualized in the creek in sufficient numbers to have largely displaced natural-origin fish. No smolt outmigration estimate exists for Whychus in part because of low captures, and no returning adult steelhead or Chinook salmon migrated into Whychus in 2014. In contrast, juvenile O. mykiss densities in Crooked River tributaries are approximately twice to three times those in Whychus; the resident population remains predominantly (91%) natural-origin, suggesting natural-origin fish are successfully competing in these tributaries while reintroduced O. mykiss smoltify and outmigrate; outmigrating steelhead and chinook smolts number in the hundreds each year; and the vast majority of returning adult steelhead and Chinook salmon are returning to the Crooked River.

Mazzacano (2015) examines six years of macroinvertebrate data to identify trends in macroinvertebrate community composition before and after extensive streamflow restoration. Individual community attributes strongly suggest a community whose composition has changed in response to changing stream conditions, reflecting a greater abundance and/or diversity of taxa that thrive in conditions characterized by cooler temperatures and lower fine sediment, despite other assessments (two indices of biological integrity [IBIs] and a predictive model) that do not indicate substantial improvements in biological conditions in Whychus Creek.

These six reports evaluate improvements in stream conditions in 2014 as measured by the status of physical and biological indicators subsequent to major streamflow, channel, and floodplain restoration, and irrigation diversion retrofits. Our intent is that these reports and the data they contain will help restoration partners understand the effectiveness (ecological outcomes) of their actions at moving the creek toward desired conditions. We hope restoration partners continue to draw from these reports to continually improve restoration implementation and monitoring in Whychus Creek.

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Whychus Creek Stream Flow

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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. In 2014, they occurred in October. 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 35 cfs in 2014. These results suggest that Whychus Creek still experiences low flows during both late spring/early summer and late summer/early fall flow, two periods when irrigation demands generally exceed water availability. Extreme flows, however, appear to be decreasing in magnitude during both of these periods. 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 and human disturbances also prevent natural disturbances from occurring. Both types of disturbance alter stream habitat (NRC 2002). Irrigation diversions along Whychus Creek diverted up to 85% 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

85- 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.

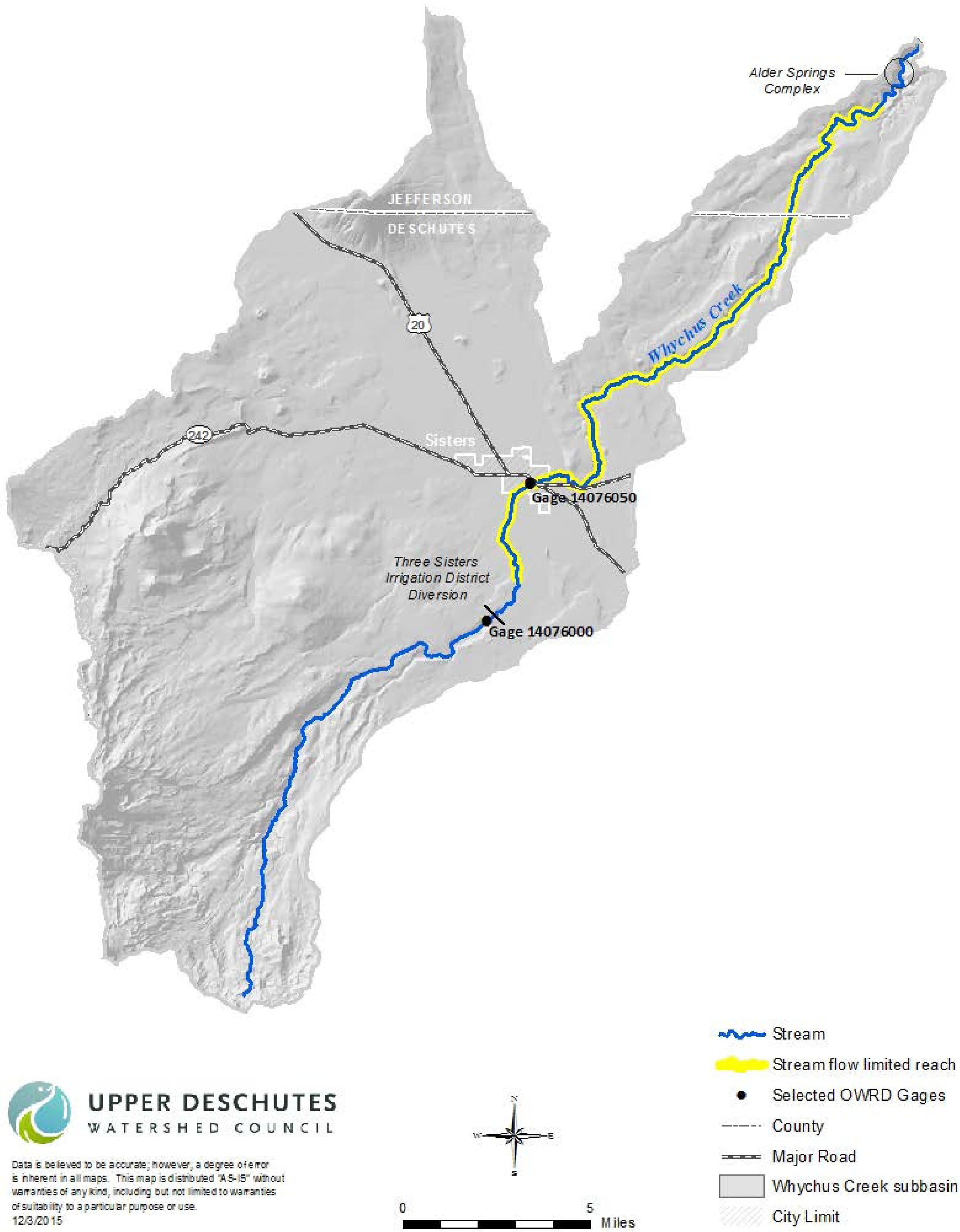

Figure 1.
Stream flow limits stream function in Whychus Creek downstream from the Three Sisters Irrigation District Diversion. Spring inputs near the mouth of Whychus Creek increase stream flow and improve conditions in the creek.

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.

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

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 2015.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 2015. Why not estimate historic stream flows at the City of Sisters over a longer time period for these analyses? Water transactions for stream flow restoration in Whychus Creek occurred during every year of the study period. Conditions through the study period are neither static nor represented by historic conditions. The period from 2000 through 2015 reflects conditions in the creek during ongoing restoration efforts.

Gage 14076050 records stream stage in Whychus Creek at Sisters, OR. The gage consists of a float-tape system that records stream stage every fifteen minutes (Burright A. Personal communication. August 24, 2009). OWRD obtains preliminary data from this gage on a near-real time basis through an automated, remote telemetry-based process. OWRD reviews this data based on their knowledge of site conditions and site-specific stage-discharge relationships. They estimate any missing values and revise any values believed to be erroneous (OWRD 2009a). OWRD reviews this data again before publishing it as daily average discharge data online. OWRD had published final data from May 18, 2000 through September 30, 2008 and from October 1, 2009 through September 30, 2011 when this report was prepared. OWRD had released provisional data from October 1, 2008 through September 30, 2009, October 1, 2011 through October 31, 2012, and from October 1, 2014 thought October 31, 2014 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 2014. 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 2014.

May Median

The DRC used spreadsheet software to determine the median daily average flow during the month of May for years 2001 through 2014. The DRC only had partial data for 2000 and did not include that data in this analysis.

August Median

The DRC used spreadsheet software to determine the median daily average flow during the month of August for years 2000 through 2014. 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 22.00 cfs in 2013 and 2014. It increased or remained constant each year except for 2005 and 2009.

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.

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

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.

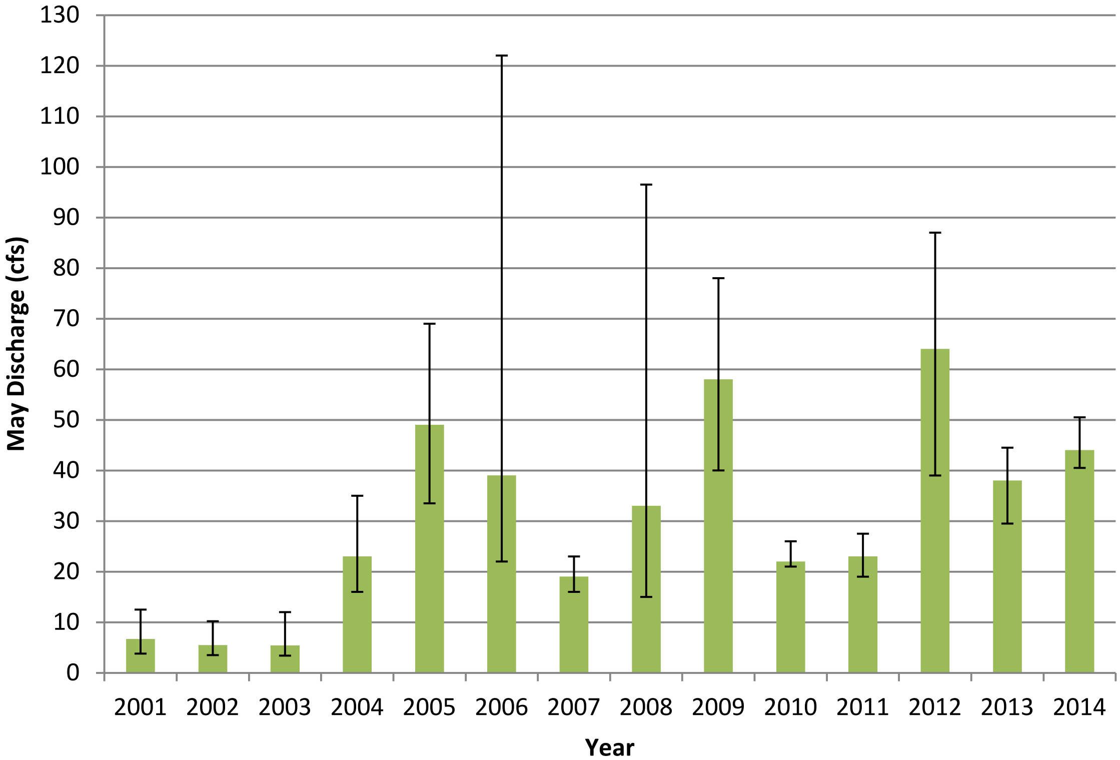

Figure 2.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of May provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of May at this location.

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.

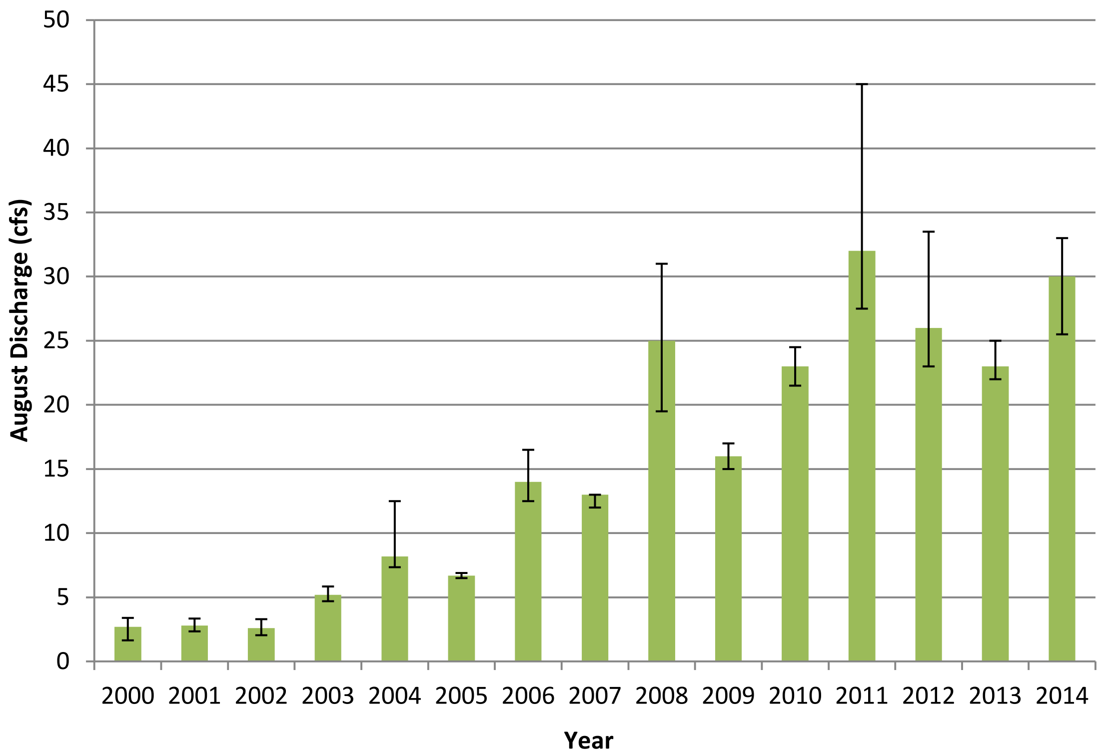

Figure 3.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of August provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of August at this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

Discussion

The analyses in this report describe stream flow conditions in Whychus Creek during a fifteen-year period of intensive restoration. They focus on the period from 2000 through 2014. 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 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.

Instream water rights awarded to the State of Oregon in the 1990s to support fish populations provide one base flow target. Median daily average flow during the month of May exceeded Oregon's 20 cfs instream water right for Whychus Creek upstream from Indian Ford Creek in ten 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 2).

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 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 low flows do not necessarily limit stream functions. The magnitude and frequency of these flows in Whychus Creek, though, suggest that low flows may limit fish populations.

The State of Oregon instream water right again provides 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, 2013 and 2014 (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 and may limit fish populations. 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.

Recommended Actions

Restoration partners have focused on restoring base flows to this historically dewatered stream system. They have operated under the assumption that base flows are critical to providing the habitat necessary to support self-sustaining populations of anadromous fish. They used, and continue to use, the instream water rights awarded to the State of Oregon as stream flow targets. Legally protected stream flows are currently approaching state instream water rights for some locations during some times of year. The reliability of these water rights varies based on water availability in Whychus Creek, leading to inter- and intra-annual variability in the low flow metrics discussed earlier. Restoration partners should continue to evaluate these low flow metrics to understand how 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. High stream flow events continue to occur before, during and after the irrigation season. Irrigation operations have reduced the magnitude of but not eliminated these events. Although describing a desired hydrograph would better inform restoration partners about the status of the stream flows in Whychus Creek, it would likely not improve the ability of restoration partners to address other hydrograph components. The absence of any storage reservoirs along Whychus Creek and the continued presence of high flow events have reduced the priority of evaluating non-base flow components of the hydrograph.

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

Acknowledgements

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

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

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Abstract

Diversion of almost 90% of summer stream flow and channelization of over 50% of the length of Whychus Creek have degraded water quality, resulting in an ODEQ listing of water quality limited since 1998. The Upper Deschutes Watershed Council monitored temperature from 1995 through 2014 at eleven sites representing diverse flow conditions in Whychus Creek. This report incorporates 2014 data to 1) evaluate the 2014 status of temperature in Whychus Creek relative to state standards for salmonid spawning, rearing and migration; 2) quantify temperature trends in relation to stream flow; and 3) update target flows projected to produce temperatures that meet state standards. Temperatures exceeded the state rearing and migration temperature standard of 18˚C at five monitoring sites in 2014 for a total of 59 days, down from 81 days in 2013 but higher than or equivalent to any other year since 2007. Temperatures exceeded the state spawning criteria for salmon and steelhead trout at six sites for 3-13 days (6-39% of data days) between January 1 and May 15 in 2014. Temperatures never exceeded the 24˚C lethal threshold for salmon and steelhead in 2014, for the fifth consecutive year. Regression of 2000-2014 temperature and flow data identified 62 cfs as the minimum flow necessary to meet the 18°C temperature standard ±2.6°C at FS Road 6360, similar to flows predicted by previous years' models and well above the 33 cfs state instream water right. Temperature results from 2014 show that the current instream water right of 33 cfs is insufficient to meet the state temperature standard and fish requirements in downstream reaches of Whychus Creek in all but the biggest water years. These results are consistent with conclusions from previous years. Continued development of creative solutions to allocation of Whychus Creek stream flow in low-water years is needed to guarantee conditions that will support the recovery of re-introduced native fish populations. 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. Whychus Creek has been listed by ODEQ under Clean Water Act Section 303(d) as water quality limited with TMDLs needed for temperature (Category 5) and categorized as having insufficient data for assessment for dissolved oxygen and pH (Category 3)(Table 1, Figure 1) since 1998.

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

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

ODEQ state temperature standards were developed to protect fish and other aquatic life in Oregon waterways (ODEQ 2009). The year-round temperature standard applied to Whychus Creek for salmon and trout rearing and migration specifies that seven-day moving average maximum (7DMAX) temperatures are not to exceed 18˚C. The 2002 303(d) list also identified Whychus Creek as not meeting the 12.8˚C state temperature standard for salmon and steelhead spawning. No subsequent 303(d) list has applied this criterion to Whychus Creek because anadromous fish were not spawning in Whychus Creek when data for these lists were collected. However, this habitat use is anticipated to resume, and the spawning temperature standard to become relevant, as steelhead and salmon reintroduced in 2007 and 2009 begin to return to the creek. The State of Oregon 1992-1994 Water Quality Standards Review (ODEQ 1995) identified 24˚C as the lethal temperature threshold for salmon and trout.

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

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

Whychus Creek is categorized as having insufficient data for assessment for dissolved oxygen and pH. UDWC analyses of dissolved oxygen data collected from 2006 to 2008 indicated that Whychus Creek met state dissolved oxygen standards for salmon and trout rearing and migration, although dissolved oxygen levels did not consistently meet state criteria for salmon and trout spawning (Jones 2010).

Because dissolved oxygen saturation is directly affected by temperature, we expect dissolved oxygen levels to track temperature trends. While observed trends in stream temperature continue to demonstrate cooling, and in the absence of other novel environmental conditions, we expect dissolved oxygen levels to improve or remain constant. Under these circumstances, temperature data are 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 stream flow restoration. Accordingly we also discontinued monitoring pH subsequent to 2009. While this report does not present dissolved oxygen or pH data, we consider the observed trends in temperature to provide a surrogate measure of water quality in Whychus Creek. For further discussion of temperature, dissolved oxygen, pH, and state standards for each parameter, refer to *Whychus Creek Water Quality Status, Temperature Trends, and Stream flow Restoration Targets* (Jones 2010).

The stream flow and habitat restoration efforts of Deschutes River Conservancy (DRC), UDWC, and restoration partners aim to improve water temperatures to meet the18˚C state standard and support sustainable anadromous and resident native fish populations by reducing warming rates and reconnecting the creek to floodplains and groundwater. DRC and restoration partners identified a stream flow target for Whychus Creek according to state instream water rights. State of Oregon March, April and May instream water rights protect 20 cfs upstream and 50 cfs downstream of Indian Ford Creek (RM 18); state water rights for June and July, and for August and September when flows are historically low, specify 20 cfs upstream and 33 cfs downstream of Indian Ford Creek. Protected water rights correspond to recommended minimum flows identified through the Oregon Method, which relates stream flow to fish habitat availability (Thompson 1972), however minimum flows identified may not be sufficient to create suitable conditions for fish or meet state temperature standards. The DRC stream flow restoration target aims to protect 33 cfs instream at Sisters City Park. Because no substantial flows enter Whychus Creek between this location and Alder Springs just below WC 001.50, the DRC target will effectively also protect 33 cfs downstream of Indian Ford Creek.

This report presents analyses of 2000-2014 temperature and flow data that: 1) evaluate the 2014 status of temperature in Whychus Creek relative to state standards for salmonid spawning, rearing and migration; 2) quantify temperature trends in relation to stream flow; and 3) update target flows projected to produce temperatures that meet state standards.

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

	Parameter	**Temperature**		**Dissolved Oxygen**		**pH**	
	Beneficial Use	Salmon & Trout Rearing & Migration	Steelhead Spawning	Salmon & Steelhead Non-Spawning	Salmon & Trout Spawning	Multiple Uses	Multiple Uses
	Season	Year Round	January 1 - May 15	Year Round	January 1 - May 15	Fall/ Winter/ Spring	Summer
	Standard	18° C	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 2014

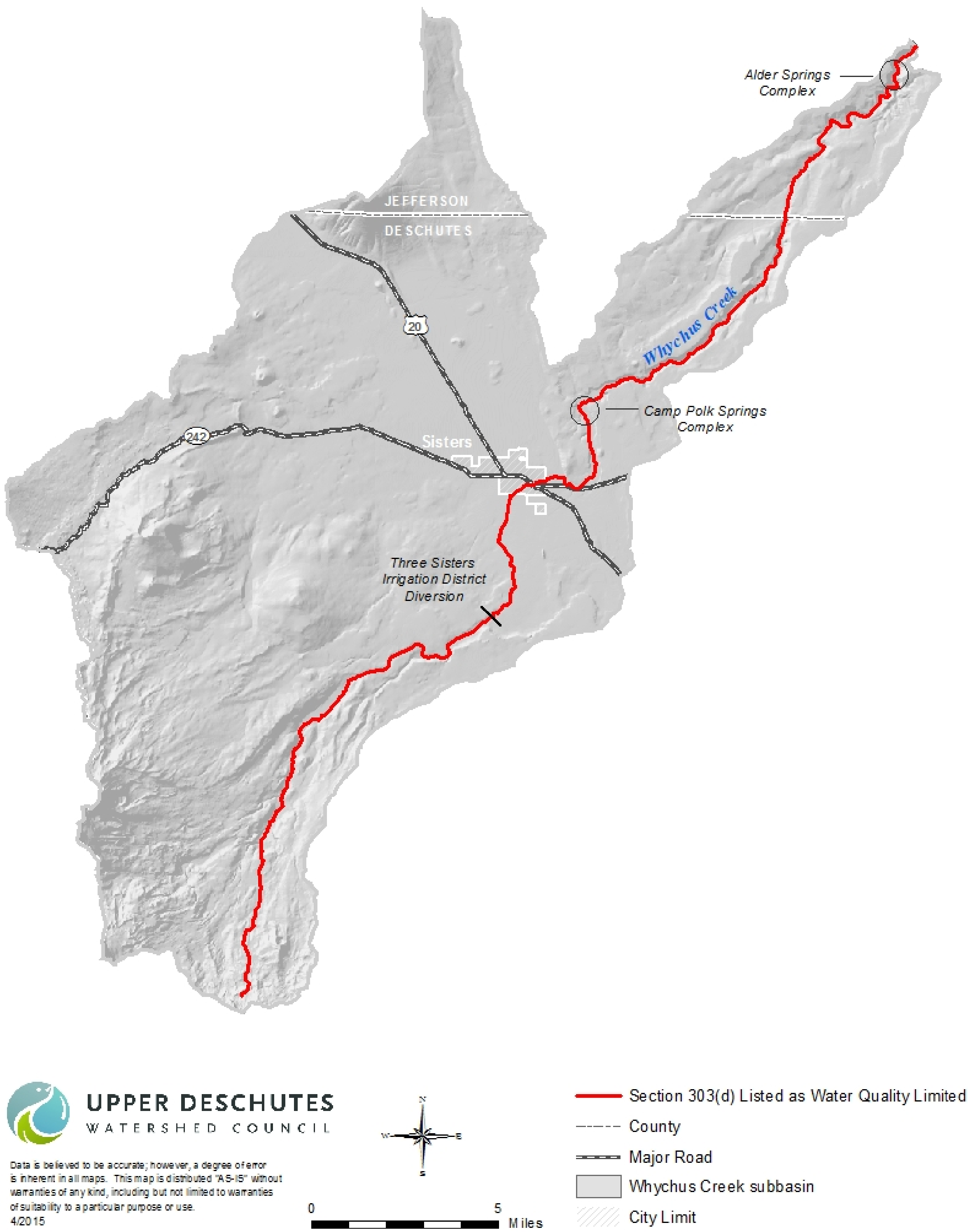


Figure 1.
Whychus Creek is listed as Water Quality Limited from river mile (RM) 0.0 to RM 40.3 under ODEQ's 2012 303(d) list. (ODEQ 2014)

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

We obtained average daily stream flow (QD) data for Whychus Creek from Oregon Water Resources Department (OWRD) gage 14076050 at the City of Sisters (OWRD 2014). This gage is located downstream from the Three Sisters Irrigation District diversion and other major irrigation diversions. We use data collected at this gage from 2000 to 2014 in this report, including some data considered by OWRD to be provisional and subject to change.

Temperature status

We evaluated 2014 seven day moving average maximum daily temperatures (7DMAX) relative to the 18˚C state temperature standard for salmonid rearing and migration and the 12.8˚C state standard for steelhead and salmon spawning according to methods described in the ODEQ *Assessment Methodology for Oregon's 2004/2006 Integrated Report on Water Quality Status* (ODEQ 2006). Because steelhead spawning season has yet to be identified for Whychus Creek, we reference the January 1 – May 15 spawning season identified for the Lower Deschutes sub-basin for evaluation of temperature relative to the 12.8˚C state standard for steelhead and salmon spawning. 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, ODFW water rights, and DRC stream flow restoration targets. We also compared the status of Whychus Creek temperatures in 2014 to temperatures from 2000-2013.

To detect changes in rates of warming at monitoring sites signaling potential emerging changes in stream conditions we calculated the rate of temperature change between sites and the average longitudinal rate of temperature change along Whychus Creek on the hottest water day of 2014. We calculated the rate of change between individual monitoring sites by dividing the difference in temperature between sites by the distance between sites. We calculated the average longitudinal rate of change as the average of all rates of change between individual monitoring sites. We defined the hottest water day as the day with the single hottest seven day moving average maximum temperature

(7DMAX). We compared the rate of change for each site 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 increases, flagging sites for further evaluation of factors that may be affecting stream temperature. Lower than average longitudinal changes in temperature highlight reaches where cooling occurs and which should be considered for protection in surface and groundwater planning.

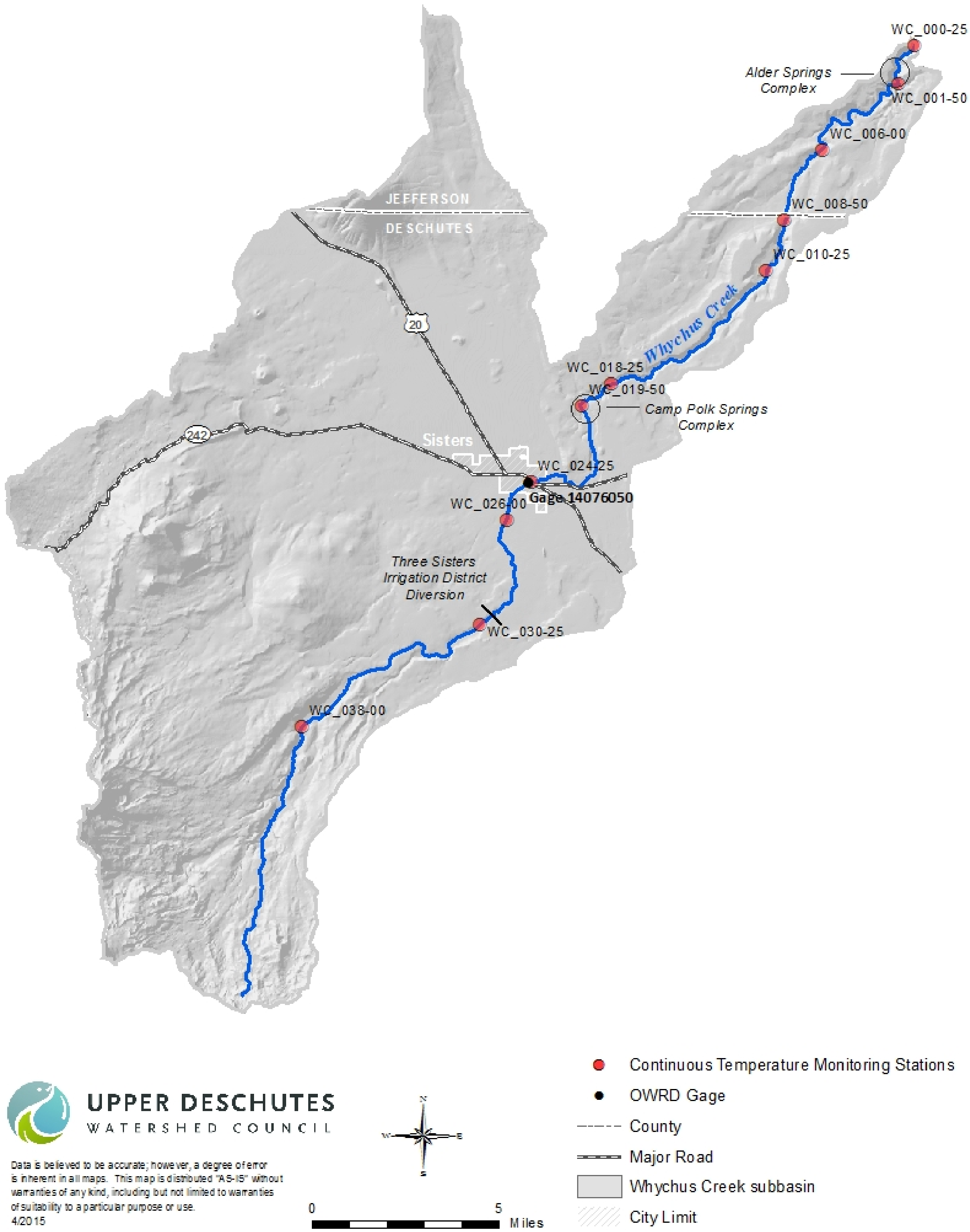


Figure 2.

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

Target Stream flow

We added 2014 data to the 2000-2013 dataset to perform a temperature-stream flow regression that refines the target stream flow 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 0006.00, with stream flow data from the OWRD gage at Sisters, to identify the stream flow 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 0006.00 represent the historically worst temperature conditions on the creek, and thus the location that is both most critically in need of and also stands to benefit the most from stream flow restoration.

We restricted data included in the regression to a one-month (30-day) interval to reduce the effect of intra-annual seasonal variation in the analysis (Helsel and Hirsch 2002) and identified July as the month during which 7DMAX temperatures most often begin to rise above 18°C (UDWC unpublished data). Daily stream flow data for all July days from 2000-2014 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 calculate mean temperatures and prediction intervals at corresponding flows for the two locations we performed a regression of temperature and flow data. The resulting equations represent the relationship between flow and temperature and can be used to estimate temperature values for the specified locations, within the evaluated time period, and within the range of flows observed.

We paired 7DMAX temperature with the corresponding natural logarithm of the average daily flow (LnQD) for each day included in the analysis. 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 represent the flow corresponding to the temperature from the first day of the 7DMAX. We calculated the natural log to hundredths to retain the precision of average daily flow data.

We regressed 7DMAX temperature on the natural log of the average daily flow for two permutations of the data and compared regression models to select the analysis that provided the most accurate fit to the data for the purpose of identifying the stream flow rate (cfs) at which stream temperature will meet or be below 18°C. We compared temperatures, prediction intervals, standard error (S) and R^2 values calculated from: 1) all 7DMAX temperature-flow pairs; and 2) all mean 7DMAX temperature-flow pairs representing the average of all temperatures observed at a given flow for all flows for which there was at least one temperature record. Using all 7DMAX temperature-flow pairs resulted in calculated temperatures within tenths of a degree of temperatures calculated from the regression of all mean 7DMAX temperature-flow pairs, with prediction intervals at most 0.4°C greater than those calculated from all mean 7DMAX temperature-flow pairs. Because differences between regression models and calculated temperatures and prediction intervals for the two datasets were minimal, within the 0.5°C accuracy of the temperature dataloggers, we used all 7DMAX temperature-flow pairs for analysis.

We used ANOVA and Akaike Information Criterion (AIC) values in R open source statistical software to determine the highest polynomial term that statistically improved the regression model (linear, quadratic, or cubic) on the basis of the p-value (ANOVA) associated with, and numeric difference between (AIC), each model. We used a normal distribution plot to evaluate the normality of the residuals of the selected model. Using the resulting regression equation for each location, we used R to calculate the predicted temperature and 95% prediction interval for all flows within the observed range (Appendix B). The 95% prediction interval (PI) is calculated as:

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

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

We compared the resulting 2000-2014 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 stream flow restoration target, and because available Heat Source scenarios assume 33 cfs at WC 024.25 and 62 cfs at WC 0006.00, we compared 2000-2014 temperature calculations to the Heat Source estimates for these flows.

Results

Temperature status

7DMAX temperatures exceeded 18 ˚C at five sites in 2014, for up to 59 days at WC 8.50 at flows from 22 to 68 cfs (Figure 4). In keeping with comparatively lower 2014 temperatures, temperatures exceeded 18˚C for fewer days than in seven previous years. Lethal temperatures (24˚C) were not recorded at any site on Whychus Creek in 2014, for the fifth year in a row, with no 2014 7DMAX temperature higher than 20.9˚C.

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 5, Figure 6). Corresponding to a moderate water year (July median average daily flow = 49 cfs), median 7DMAX July stream temperature at Sisters City Park was lower in 2014 than in 2013 and seven other of fourteen years for which July temperature data were available (Figure 5). 7DMAX temperature at this site remained well below 18˚C throughout July. Median July temperature trends at Road 6360 tracked those at Sisters City Park, with the 2014 median lower than 2013 and lower than or equivalent to seven of eleven years for which July data were available at this site.

Continuous temperature data were available for 31-49 days of the January 1-May 15 steelhead and salmon spawning season, varying by site. Temperatures recorded for these dates exceeded the 12.8˚C spawning habitat requirement and potential state standard at six sites over 3 to 13 days from the first March date for which data were available to May 15, representing 6-39% of spawning season days for which data were available.

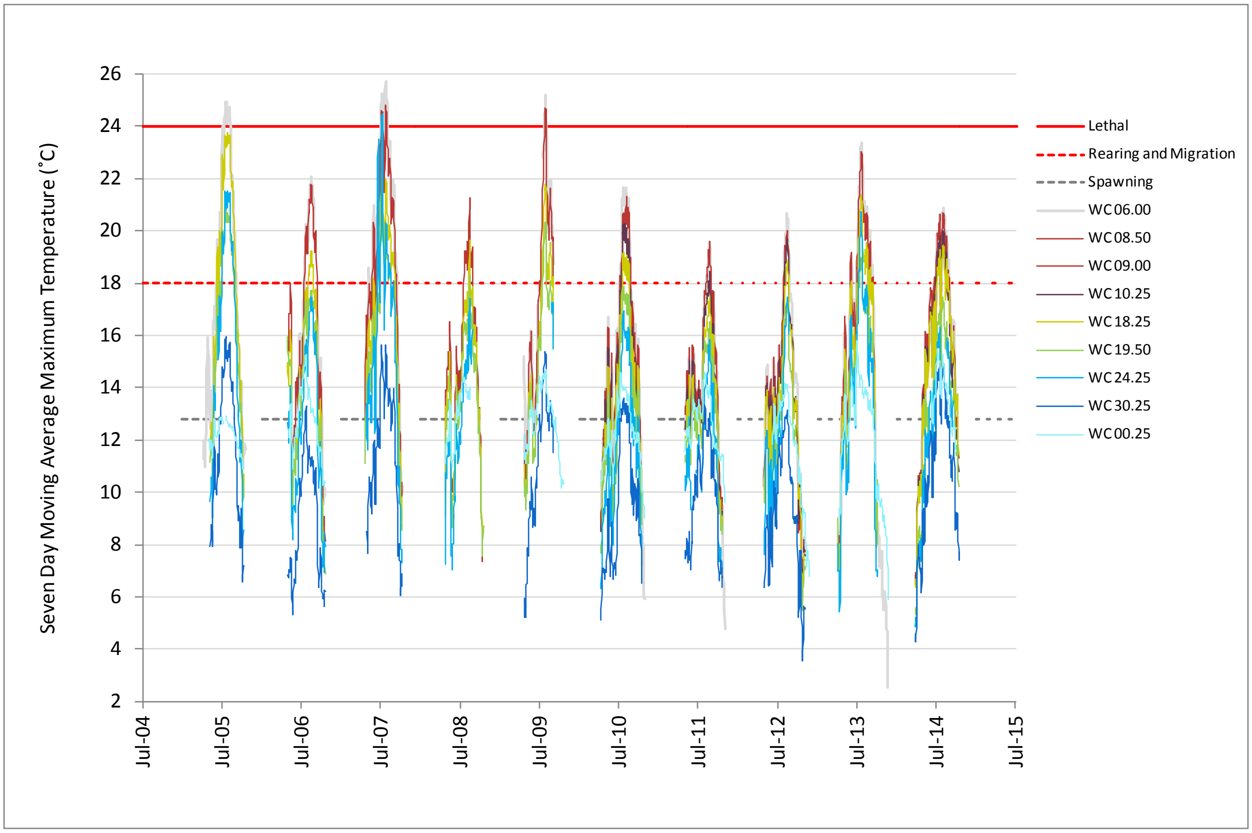
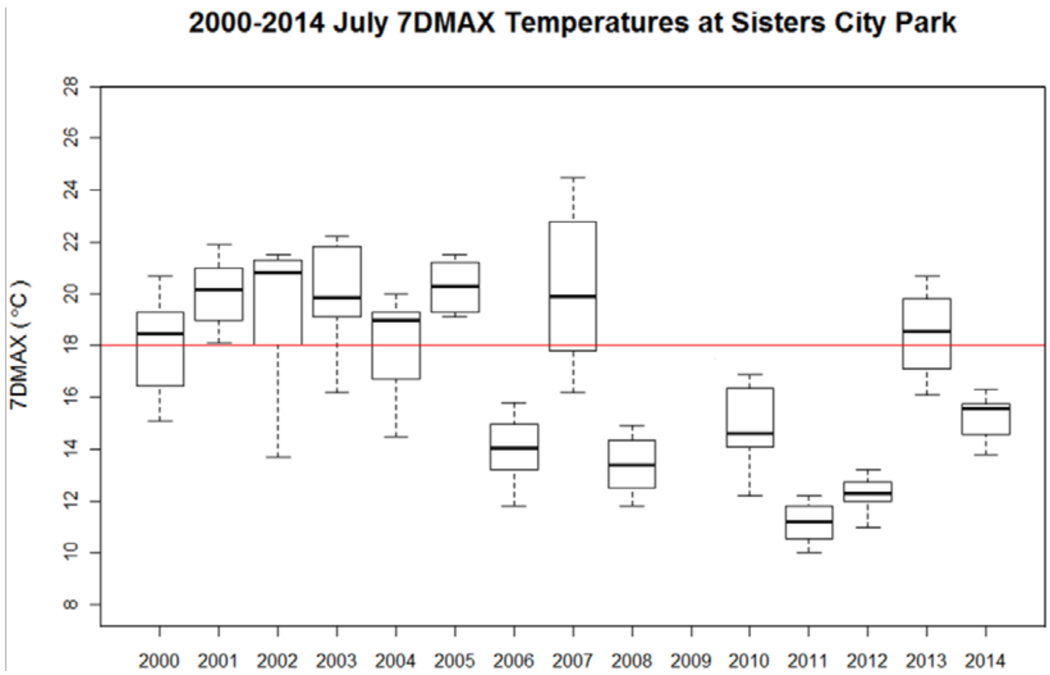


Figure 4.
7DMAX temperatures from 2004 to 2014 at nine Whychus Creek monitoring sites. Stream temperature exceeded the 18˚C state standard at five sites in 2014, from river mile 6 (WC 006.00) to river mile 18.25 (WC 018.25)

a



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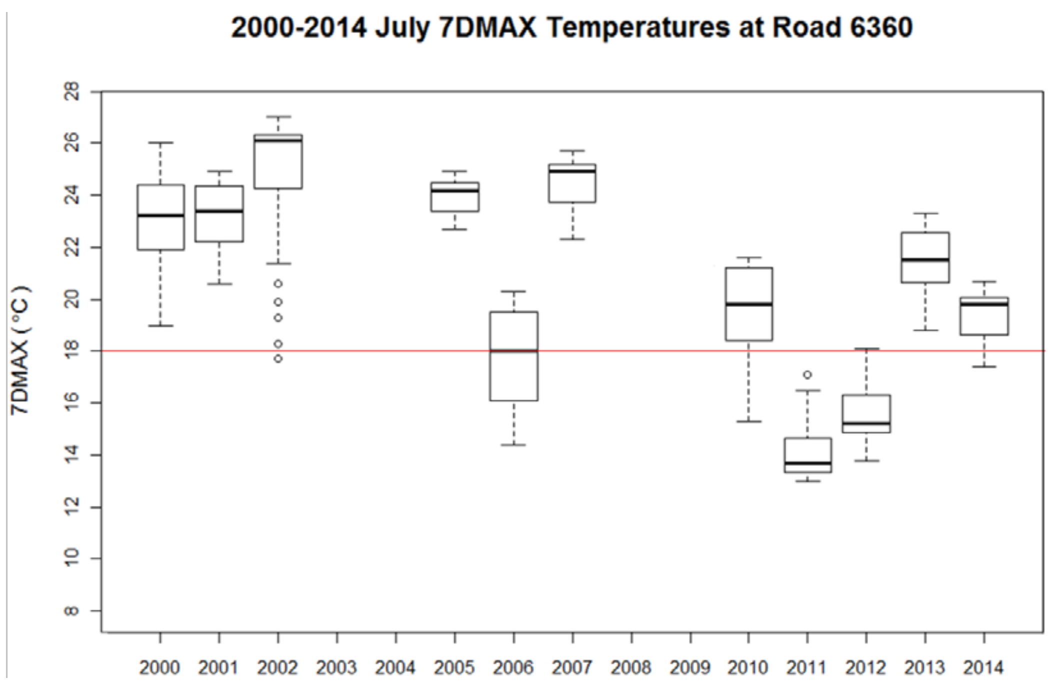


Figure 5.
July mean 7DMAX temperatures at a) Sisters City Park (WC 024.25) and b) Road 6360 (WC 0006.00) have fallen considerably since 2007 following significant gains in stream flow restoration. 2014 July temperatures were attenuated by moderately high stream flow (July median QD = 49 cfs). As in all but a few years, July 2014 temperatures at Road 6360 exceeded 18°C.

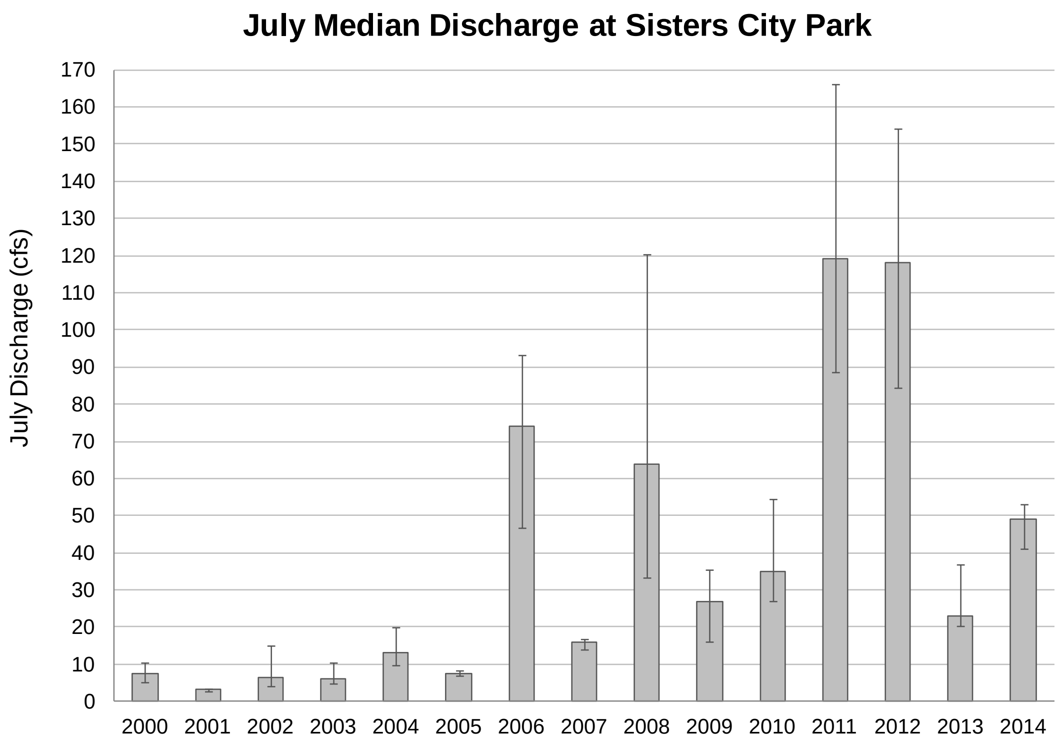


Figure 6.
July median stream flow from 2000 to 2014 corresponded closely to median temperatures at Sisters City Park and Road 6360.

The hottest water day in 2014 occurred on August 3rd, with stream temperature warming by 0.03˚C per mile on average (Figure 7). Above-average warming and cooling occurred at the same sites as in previous years. Since 2012 the greatest increase over the average rate of change has occurred along the reconstructed channel at Camp Polk; however, although the average temperature difference between WC 019.50, upstream of Camp Polk, and WC 018.25, downstream of Camp Polk, increased pre-project (2007-2011) to post-project (2012-2014), that difference is explained by a decrease in the upstream average temperature, at WC 019.50, rather than an increase in the downstream average temperature, at WC 018.25 (data not shown). Cooling between Sisters City Park and WC 019.50 suggests a potential increased cooling effect of spring water inflows upstream of WC 019.50 documented in thermal infrared data (Watershed Sciences 2008) and might signal increased flow from these springs, possibly associated with elevation of groundwater and floodplain reconnection following channel and floodplain restoration in Camp Polk Meadow. The -3.8˚C rate of temperature change from Alder Springs to the mouth of Whychus Creek was dramatically lower than the average rate of change, reflecting the cooling effect of springs complex flows.

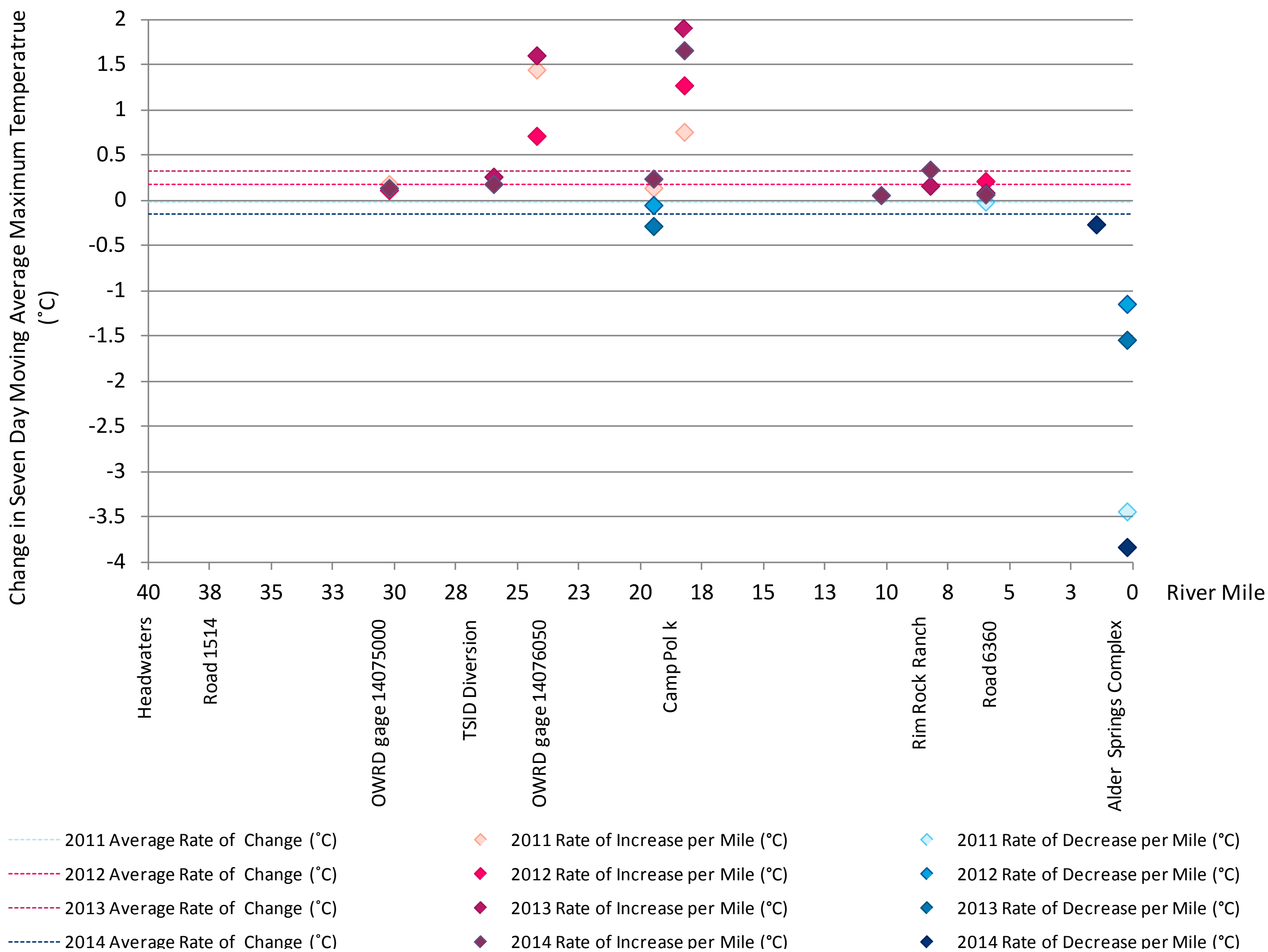


Figure 7.
Longitudinal rates of temperature change in Whychus Creek in 2011-2014 flag areas where higher than average rates of warming are occurring, as in the Camp Polk reach (RM 18.25 – 19.5) , as well as reaches where cooling is occurring.

Target stream flow

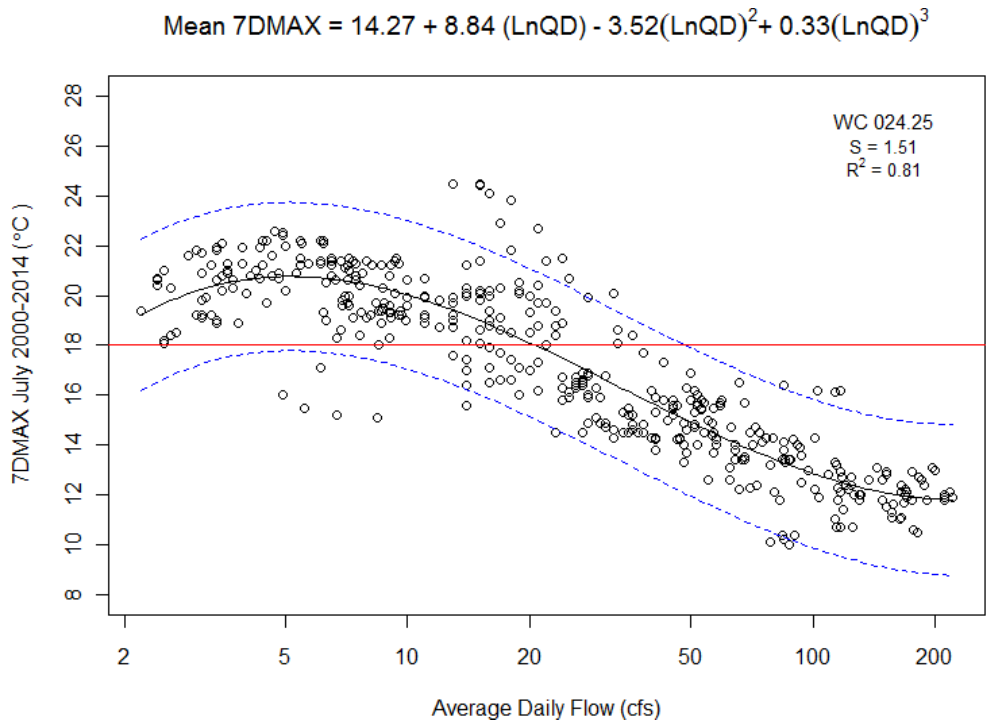
The cubic model was statistically better than lower-order models for both WC 024.25 and WC 0006.00 data (Table 3, Figure 8). Temperatures calculated from the WC 024.25 regression model suggest that 19 cfs (2.94 LnQD) was the minimum stream flow resulting in a mean 7DMAX temperature at or below 18°C (± 2.1°C) given temperatures observed from July 2000-2014 at Sisters City Park (Appendix B). The existing 33 cfs restoration target resulted in a mean 7DMAX temperature of 16.2°C ± 2.1°C at this location. The 2000-2014 estimate for Sisters City Park is somewhat higher than the 2008 Heat Source model estimate of 15°C ± 1°C at 33 cfs at the ODFW gage at Sisters City Park (Watershed Sciences and MaxDepth Aquatics 2008).

The cubic regression model for the temperature-flow relationship at Road 6360 (WC 0006.00) derived from 2000-2014 temperature and flow data estimates 62 cfs to be the minimum stream flow that will achieve a mean 7DMAX temperature of 18.0°C ± 2.6°C. According to this model the target stream flow of 33 cfs below Indian Ford Creek is projected to produce a mean 7DMAX temperature of 20.5°C ± 2.6 at Road 6360, above the 18°C state standard and above the 19.5°C threshold at which Chinook salmon growth has been shown to slow. The 2000-2014 cubic regression model estimate of 18.0°C ± 2.6°C at 62 cfs is slightly lower than the Heat Source model estimate of 18.5°C ± 1°C at 62 cfs at Road 6360.

Table 3.
The cubic regression model provided the best fit to 2000-2014 temperature-flow data for both WC 024.25 and WC 0006.00 data. Temperatures calculated using the corresponding regression equations are expected to be the most accurate of the possible regression models.

2000-2014 Regression	Equation	df	S	adjusted R^2	ANOVA p-value	AIC value
	WC 24.25 (n=427)					
Linear	24.7614 - 2.452(LnQD)	425	1.71	0.75	--	459.84
Quadratic	21.4414 + 0.0458(LnQD) - 0.3985$(LnQD)^2$	424	1.61	0.78	0.00	410.91
Cubic	**14.2664 + 8.8378(LnQD) - 3.5193$(LnQD)^2$ + 0.3330$(LnQD)^3$**	**423**	**1.51**	**0.81**	**0.00**	**356.17**
	WC 0006.00 (n=354)					
Linear	29.1646 - 2.6595(LnQD)	352	1.98	0.73	--	484.63
Quadratic	22.5205 + 2.4047 (LnQD) - 0.8163$(LnQD)^2$	351	1.59	0.83	0.00	328.87
Cubic	**17.068 + 9.2859(LnQD) - 3.3024$(LnQD)^2$ + 0.2689$(LnQD)^3$**	**350**	**1.52**	**0.84**	**0.00**	**302.10**

a



b

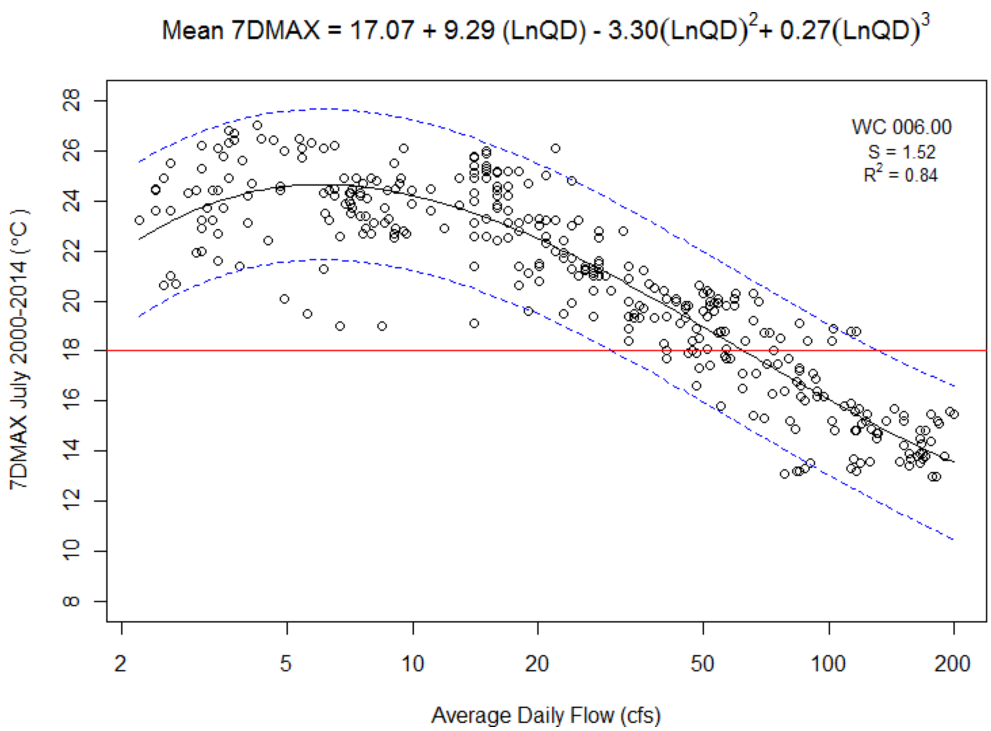


Figure 8.
Regression models fitted to temperature-flow data from July 2000-2014 describe the relationship between temperature and flow observed at a) Sisters City Park (WC 024.25), and b) Road 6360 (WC 0006.00), fitted with cubic regression model trendlines. We used the corresponding regression equations to calculate the mean temperature predicted to occur at a given flow.

Discussion

Temperature status

Temperatures in excess of 18˚C over a prolonged duration along the length of Whychus Creek in 2014 support the ODEQ 2012 303(d) Category 5 listing of Whychus Creek as water quality limited (ODEQ 2014). Because the state standard is biologically based, we can surmise that temperature conditions above 18˚C compromised habitat suitability for rearing and migrating trout and salmon in Whychus Creek, at a minimum through Camp Polk and reaches downstream, in 2014. Seven day moving average maximum temperatures also indicate marginal conditions for spawning trout and salmon. 7DMAX temperatures remained below the 24˚C lethal threshold in 2014.

Temperature data from 2000 through 2014 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 2008 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. 2014 July temperatures attenuated from 2013, corresponding to higher July flows represented by a July median flow twice the 2013 median.

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 has increased the minimum flow delivered instream, resulting in higher monthly median flows that reflect consistently higher average daily flows, which in turn correspond to lower observed temperatures.

Target stream flow

The state water right for Whychus Creek protects 20 cfs instream above Indian Ford Creek, between RM 20 and RM 21, and 33 cfs downstream of Indian Ford Creek. Because no additional flows enter Whychus Creek between the headwaters and Indian Ford Creek, 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 0006.00) 2000-2014 temperature and flow data indicate a minimum flow of 62 cfs is necessary to achieve stream temperatures of 18°C±2.6°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 may 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.

Conclusions

The temperature-flow relationship described by fourteen years of data indicates 62 cfs is the minimum flow at which stream temperatures will meet 18°C at Road 6360. 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 and above 62 cfs will most reliably result in temperatures that create suitable stream conditions for rearing and migrating trout and salmon. Although 62 cfs may not currently be a feasible restoration target, these data nonetheless provide a benchmark for stream flow restoration and, importantly, show the 33 cfs state water right to be far short of the flows needed to meet the state temperature standard or provide suitable conditions for fish. Small gains in stream flow restoration that result in similarly small reductions in temperature are nonetheless likely to improve habitat conditions for some fish in some locations, for example by providing adequate flow for steelhead outmigration, increasing channel margin habitat by increasing channel width, and creating pools and cover for resident redband.

Our results show that higher stream flow achieved in part through stream flow restoration results in lower temperatures and better stream conditions for re-introduced salmon and trout, 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.

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APPENDIX A Whychus Creek continuous temperature monitoring stations 1995-2014. TE = Temperature status, ST = Stream flow target.

Station ID	Description	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
WC 038.00	Road 1514				TE				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	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	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	TE
WC 018.25	d/s end of Camp Polk							TE		TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE	TE
WC 010.25	u/s end of Rimrock Ranch															TE	TE	TE	TE	TE	TE
WC 009.00	Rimrock Ranch												TE	TE	TE						
WC 008.50	d/s end of Rimrock Ranch															TE	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	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	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	TE

APPENDIX B Temperatures at given flows.

Whychus Creek at Sister's City Park (WC 024.25) at flows from 2 to 221 CFS.

Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)
2	19.1	2.3	57	14.4	2.1	112	12.6	2.1	167	12.0	2.1
3	20.2	2.2	58	14.4	2.1	113	12.6	2.1	168	12.0	2.1
4	20.6	2.1	59	14.3	2.1	114	12.6	2.1	169	12.0	2.1
5	20.6	2.1	60	14.3	2.1	115	12.6	2.1	170	12.0	2.1
6	20.5	2.1	61	14.2	2.1	116	12.5	2.1	171	11.9	2.1
7	20.4	2.1	62	14.2	2.1	117	12.5	2.1	172	11.9	2.1
8	20.2	2.1	63	14.1	2.1	118	12.5	2.1	173	11.9	2.1
9	20.0	2.1	64	14.1	2.1	119	12.5	2.1	174	11.9	2.1
10	19.8	2.1	65	14.0	2.1	120	12.5	2.1	175	11.9	2.1
11	19.5	2.1	66	14.0	2.1	121	12.5	2.1	176	11.9	2.1
12	19.3	2.1	67	13.9	2.1	122	12.4	2.1	177	11.9	2.1
13	19.1	2.1	68	13.9	2.1	123	12.4	2.1	178	11.9	2.1
14	18.9	2.1	69	13.8	2.1	124	12.4	2.1	179	11.9	2.2
15	18.7	2.1	70	13.8	2.1	125	12.4	2.1	180	11.9	2.2
16	18.6	2.1	71	13.8	2.1	126	12.4	2.1	181	11.9	2.2
17	18.4	2.1	72	13.7	2.1	127	12.4	2.1	182	11.9	2.2
18	18.2	2.1	73	13.7	2.1	128	12.4	2.1	183	11.9	2.2
19	18.0	2.1	74	13.6	2.1	129	12.3	2.1	184	11.9	2.2
20	17.9	2.1	75	13.6	2.1	130	12.3	2.1	185	11.9	2.2
21	17.7	2.1	76	13.6	2.1	131	12.3	2.1	186	11.9	2.2
22	17.6	2.1	77	13.5	2.1	132	12.3	2.1	187	11.9	2.2
23	17.4	2.1	78	13.5	2.1	133	12.3	2.1	188	11.9	2.2
24	17.3	2.1	79	13.5	2.1	134	12.3	2.1	189	11.9	2.2
25	17.2	2.1	80	13.4	2.1	135	12.3	2.1	190	11.9	2.2
26	17.0	2.1	81	13.4	2.1	136	12.3	2.1	191	11.9	2.2
27	16.9	2.1	82	13.4	2.1	137	12.2	2.1	192	11.9	2.2
28	16.8	2.1	83	13.3	2.1	138	12.2	2.1	193	11.8	2.2
29	16.7	2.1	84	13.3	2.1	139	12.2	2.1	194	11.8	2.2
30	16.5	2.1	85	13.3	2.1	140	12.2	2.1	195	11.8	2.2
31	16.4	2.1	86	13.2	2.1	141	12.2	2.1	196	11.8	2.2
32	16.3	2.1	87	13.2	2.1	142	12.2	2.1	197	11.8	2.2
33	16.2	2.1	88	13.2	2.1	143	12.2	2.1	198	11.8	2.2
34	16.1	2.1	89	13.1	2.1	144	12.2	2.1	199	11.8	2.2
35	16.0	2.1	90	13.1	2.1	145	12.2	2.1	200	11.8	2.2
36	15.9	2.1	91	13.1	2.1	146	12.1	2.1	201	11.8	2.2
37	15.8	2.1	92	13.1	2.1	147	12.1	2.1	202	11.8	2.2
38	15.7	2.1	93	13.0	2.1	148	12.1	2.1	203	11.8	2.2
39	15.7	2.1	94	13.0	2.1	149	12.1	2.1	204	11.8	2.2
40	15.6	2.1	95	13.0	2.1	150	12.1	2.1	205	11.8	2.2
41	15.5	2.1	96	13.0	2.1	151	12.1	2.1	206	11.8	2.2
42	15.4	2.1	97	12.9	2.1	152	12.1	2.1	207	11.8	2.2
43	15.3	2.1	98	12.9	2.1	153	12.1	2.1	208	11.8	2.2
44	15.3	2.1	99	12.9	2.1	154	12.1	2.1	209	11.8	2.2
45	15.2	2.1	100	12.9	2.1	155	12.1	2.1	210	11.8	2.2
46	15.1	2.1	101	12.8	2.1	156	12.1	2.1	211	11.8	2.2
47	15.0	2.1	102	12.8	2.1	157	12.0	2.1	212	11.8	2.2
48	15.0	2.1	103	12.8	2.1	158	12.0	2.1	213	11.8	2.2
49	14.9	2.1	104	12.8	2.1	159	12.0	2.1	214	11.8	2.2
50	14.8	2.1	105	12.7	2.1	160	12.0	2.1	215	11.8	2.2
51	14.8	2.1	106	12.7	2.1	161	12.0	2.1	216	11.8	2.2
52	14.7	2.1	107	12.7	2.1	162	12.0	2.1	217	11.8	2.2
53	14.6	2.1	108	12.7	2.1	163	12.0	2.1	218	11.8	2.2
54	14.6	2.1	109	12.7	2.1	164	12.0	2.1	219	11.8	2.2
55	14.5	2.1	110	12.6	2.1	165	12.0	2.1	220	11.8	2.2
56	14.5	2.1	111	12.6	2.1	166	12.0	2.1	221	11.8	2.2

Whychus Creek at Road 6360 (WC 0006.00) at flows from 2 to 201 cfs

Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)	Flow (cfs)	Mean Temp (7DMAX)	PI (±)
2	22.3	2.8	57	18.3	2.6	112	15.6	2.6	167	14.2	2.6
3	23.7	2.6	58	18.3	2.6	113	15.6	2.6	168	14.2	2.6
4	24.2	2.6	59	18.2	2.6	114	15.6	2.6	169	14.2	2.6
5	24.4	2.6	60	18.1	2.6	115	15.5	2.6	170	14.1	2.6
6	24.4	2.6	61	18.1	2.6	116	15.5	2.6	171	14.1	2.6
7	24.3	2.6	62	18.0	2.6	117	15.5	2.6	172	14.1	2.6
8	24.2	2.6	63	17.9	2.6	118	15.4	2.6	173	14.1	2.6
9	24.1	2.6	64	17.9	2.6	119	15.4	2.6	174	14.1	2.6
10	23.9	2.6	65	17.8	2.6	120	15.4	2.6	175	14.0	2.6
11	23.7	2.6	66	17.7	2.6	121	15.4	2.6	176	14.0	2.6
12	23.5	2.6	67	17.7	2.6	122	15.3	2.6	177	14.0	2.6
13	23.4	2.6	68	17.6	2.6	123	15.3	2.6	178	14.0	2.6
14	23.2	2.6	69	17.6	2.6	124	15.3	2.6	179	14.0	2.6
15	23.0	2.6	70	17.5	2.6	125	15.2	2.6	180	13.9	2.6
16	22.8	2.6	71	17.4	2.6	126	15.2	2.6	181	13.9	2.6
17	22.7	2.6	72	17.4	2.6	127	15.2	2.6	182	13.9	2.6
18	22.5	2.6	73	17.3	2.6	128	15.1	2.6	183	13.9	2.6
19	22.4	2.6	74	17.3	2.6	129	15.1	2.6	184	13.9	2.6
20	22.2	2.6	75	17.2	2.6	130	15.1	2.6	185	13.9	2.6
21	22.0	2.6	76	17.2	2.6	131	15.1	2.6	186	13.8	2.6
22	21.9	2.6	77	17.1	2.6	132	15.0	2.6	187	13.8	2.6
23	21.7	2.6	78	17.1	2.6	133	15.0	2.6	188	13.8	2.6
24	21.6	2.6	79	17.0	2.6	134	15.0	2.6	189	13.8	2.6
25	21.5	2.6	80	17.0	2.6	135	14.9	2.6	190	13.8	2.6
26	21.3	2.6	81	16.9	2.6	136	14.9	2.6	191	13.8	2.6
27	21.2	2.6	82	16.9	2.6	137	14.9	2.6	192	13.7	2.6
28	21.1	2.6	83	16.8	2.6	138	14.9	2.6	193	13.7	2.6
29	20.9	2.6	84	16.8	2.6	139	14.8	2.6	194	13.7	2.6
30	20.8	2.6	85	16.7	2.6	140	14.8	2.6	195	13.7	2.6
31	20.7	2.6	86	16.7	2.6	141	14.8	2.6	196	13.7	2.6
32	20.6	2.6	87	16.6	2.6	142	14.8	2.6	197	13.7	2.6
33	20.5	2.6	88	16.6	2.6	143	14.7	2.6	198	13.6	2.6
34	20.4	2.6	89	16.5	2.6	144	14.7	2.6	199	13.6	2.6
35	20.2	2.6	90	16.5	2.6	145	14.7	2.6	200	13.6	2.6
36	20.1	2.6	91	16.4	2.6	146	14.7	2.6	201	13.6	2.6
37	20.0	2.6	92	16.4	2.6	147	14.6	2.6			
38	19.9	2.6	93	16.4	2.6	148	14.6	2.6			
39	19.8	2.6	94	16.3	2.6	149	14.6	2.6			
40	19.7	2.6	95	16.3	2.6	150	14.6	2.6			
41	19.6	2.6	96	16.2	2.6	151	14.5	2.6			
42	19.5	2.6	97	16.2	2.6	152	14.5	2.6			
43	19.4	2.6	98	16.2	2.6	153	14.5	2.6			
44	19.4	2.6	99	16.1	2.6	154	14.5	2.6			
45	19.3	2.6	100	16.1	2.6	155	14.5	2.6			
46	19.2	2.6	101	16.0	2.6	156	14.4	2.6			
47	19.1	2.6	102	16.0	2.6	157	14.4	2.6			
48	19.0	2.6	103	16.0	2.6	158	14.4	2.6			
49	18.9	2.6	104	15.9	2.6	159	14.4	2.6			
50	18.8	2.6	105	15.9	2.6	160	14.3	2.6			
51	18.8	2.6	106	15.9	2.6	161	14.3	2.6			
52	18.7	2.6	107	15.8	2.6	162	14.3	2.6			
53	18.6	2.6	108	15.8	2.6	163	14.3	2.6			
54	18.5	2.6	109	15.7	2.6	164	14.3	2.6			
55	18.5	2.6	110	15.7	2.6	165	14.2	2.6			
56	18.4	2.6	111	15.7	2.6	166	14.2	2.6			

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, Whychus Falls, to five. A third inventoried barrier was deteRMined not to present a barrier to fish passage. No new fish passage projects were completed in 2012. In 2013, dam removal and channel restoration were completed at a fourth former barrier to fish passage, connecting an additional two reaches and leaving three reaches below the natural barrier. The last concrete dam on Whychus Creek was removed in 2014, increasing miles accessible from the mouth of the creek by three, to 26.8, and leaving only one 11.5-mile reach inaccessible between the final barrier and Whychus Falls. UDWC continues to actively engage water rights holders to provide passage at the final passage barrier by 2017. Removal of this barrier will provide access to 11.5 additional miles of habitat and restore connectivity along the entire length of Whychus Creek historically accessible to resident and anadromous species.

Introduction

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

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

Monitoring the river miles of habitat opened to resident and anadromous fish through barrier removal provides a measure of stream habitat connectivity. Fish population data 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 as of 2009. We calculated the number of fragmented reaches created by the barriers and the number of stream miles for each reach. This report presents the status of fish passage and stream connectivity at the close of 2014.

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, leaving a total of four fragmented reaches 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. This barrier, the last concrete dam in Whychus Creek, was removed in September, 2014.

Table 1.
Passage barrier specifications and status as of 2014. UDWC collected baseline data on passage barriers in Whychus Creek in 2009. Data that were not available were estimated based on OWRD surveys completed in 2002.

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

[1] Water elevation difference above and below the hydraulic jump. **Must be ≤ 6 inches**
[2] Depth of water in plunge pool downstream of hydraulic jump. **Must be ≥ 24 inches**

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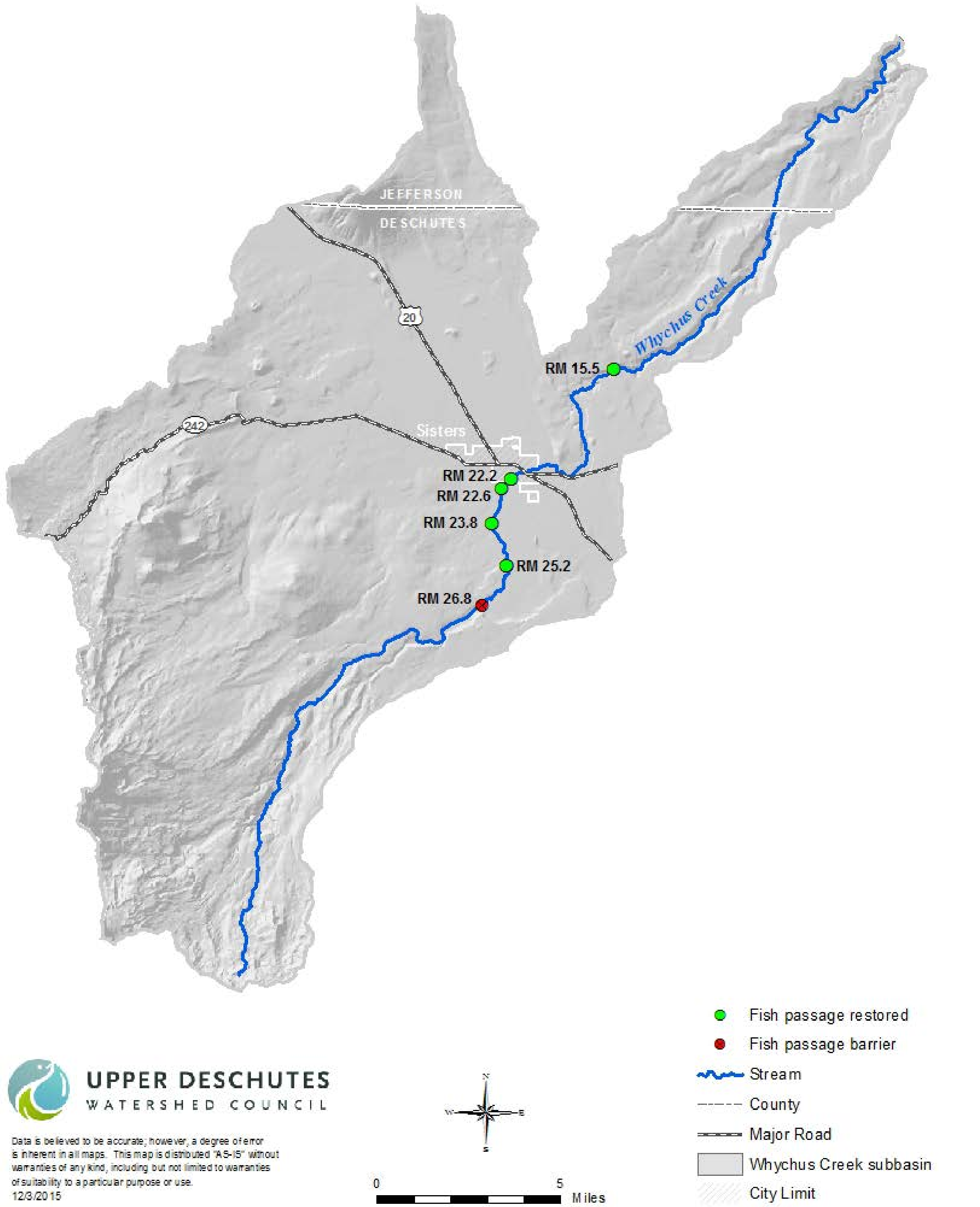

Figure 1.
In 2009, six barriers impaired stream connectivity on Whychus Creek between river miles 15.5 and 26.8. Over five years, from 2009 to 2014, fish passage was restored or found to be intact at five of these barriers. UDWC and restoration partners aim to provide passage at the one remaining barrier by 2017.

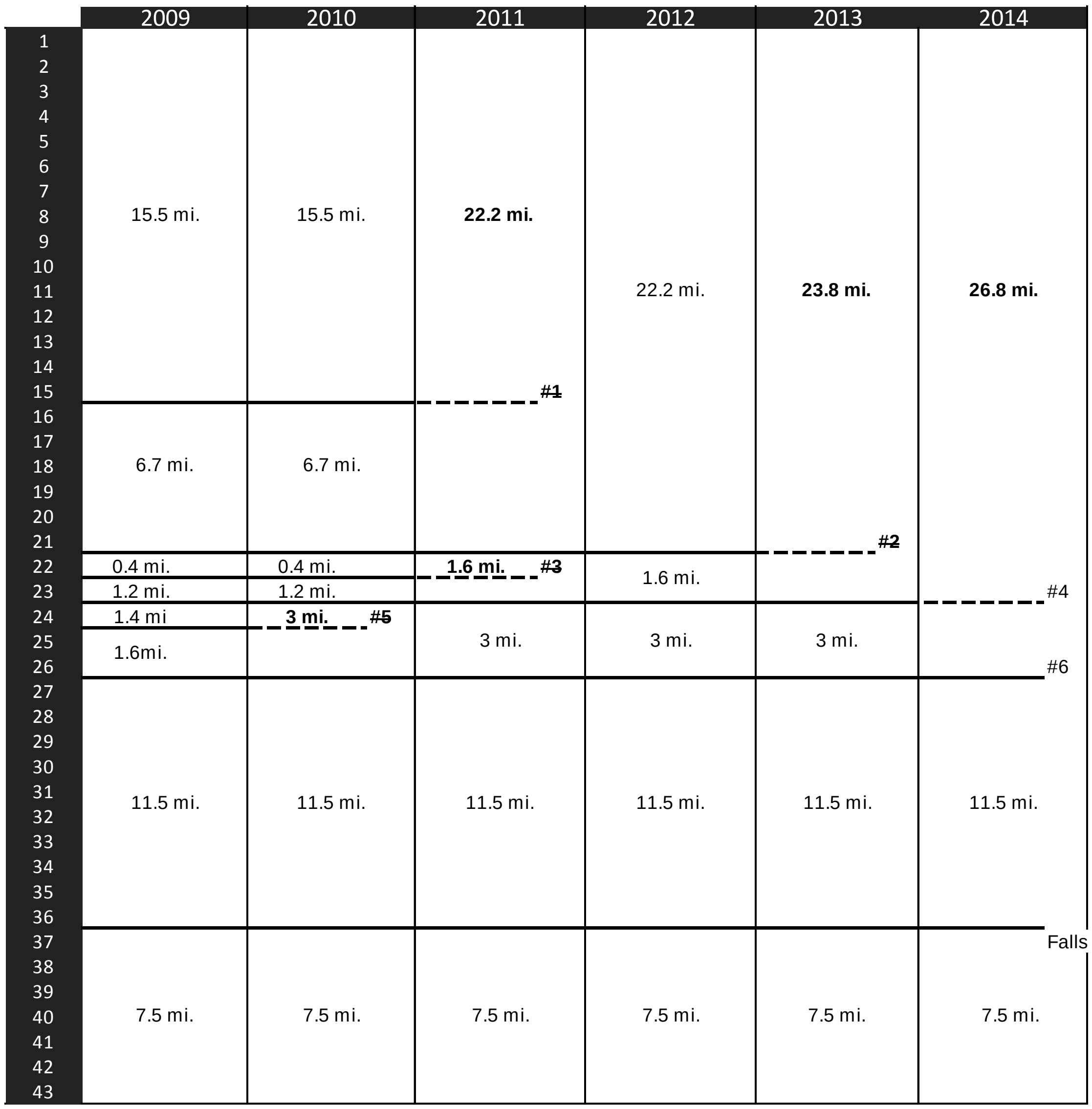

Figure 2.

Only one barrier to fish passage remains as of 2014, reducing the number of fragmented reaches from a baseline of seven to two: 26.8 miles accessible from the mouth of the creek, and 11.5 miles above the last remaining barrier. Whychus Creek Falls, located between river miles 36 and 37, is the downstream-most natural barrier to fish passage.

Discussion

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

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Fish Entrainment Potential in Whychus Creek

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Abstract

The Upper Deschutes Watershed Council (UDWC) selected fish entrainment potential as an indicator of restoration effectiveness on Whychus Creek, 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 to 2014, restoration partners screened four diversions, while water rights transactions reduced the total diversion rate by 27.24 cfs. In 2013 and 2014, 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 87% from 2009 to 2014, from 192.89 to 22.45 cfs. Potential for fish entrainment in irrigation diversions has thus been substantially reduced. UDWC and restoration partners will continue to engage with water rights holders and landowners to eliminate risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

Introduction

UDWC selected fish entrainment potential as an indicator of restoration effectiveness on Whychus Creek. UDWC described fish entrainment potential by the presence and location of irrigation diversions lacking state and federally approved fish screens. Irrigation diversions can create two types of problems for fish. First, they potentially block upstream and downstream fish passage. Second, unscreened diversions divert fish almost as effectively as they divert water. This technical report updates the potential for fish entrainment at irrigation diversions in Whychus Creek at the close of 2014.

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 in 2012 to a new low of 32.25. 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 was designed to, among other objectives, remove the historic diversion at Pine Meadow Ranch and replace it with a pump outfitted with a fish screen. The Whychus Floodplain Restoration Plan was completed and peer reviewed in 2013, and installation of the new pump and screen was implemented in 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 2014 conversations with water managers and water rights holders to address fish screening at Diversion No. 2 were ongoing. Screening or decommissioning the three diversions currently in various stages of planning will reduce the total number of unscreened diversions to 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, one diversion was decommissioned, and one was replaced with a pump, between 2009 and 2014, leaving six diversions unscreened.

Diversion ID	Baseline Sampling Date	River Mile	Diversion Type	2009 Associated Diversion Rate (cfs)	Screen Present at Baseline Inventory	Screen opening size (inches)	Met State & Federal Criteria at Baseline Inventory	2014 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	4.00	Apr-14	Yes	Sokol
No. 7	8/28/2009	20.90	Gravity	1.12	No	N/A	No	0.00	Oct-09	Yes	Leithauser
No. 8	8/28/2009	18.65	Pump	0.07	Yes	1/4	No	0.07		No	No. 9 on OWRD List
No. 9	8/28/2009	18.15	Pump	0.38	Yes	1/4	No	0.38		No	Bradley
No. 10	8/28/2009	17.50	Pump	0.45	Yes	3/32	Yes	0.45	Aug-09	Yes	Deggendorfer
No. 11	9/30/2002	14.75	Pump	0.05	Yes	No Data	No	0.05		Unknown	Meyer. Screened, but unknown whether screen meets criteria.
No. 12	9/24/2002	11.20	Gravity	0.68	No	N/A	No	0.68		No	Remund. Not diverted in 2014; will be removed or screened within Whychus Canyon project
No. 13	9/24/2002	9.25	Gravity	0.60	No	N/A	No	0.60		No	Baker.0.24 cfs leased instream in 2014; diversion will be removed or screened within Whychus Canyon project
			Baseline Diversion Total	**193.34**			**2014 Diversion Total**	**166.38**			
			Baseline Unscreened Total	**192.89**			**2014 Unscreened Total**	**22.45**			

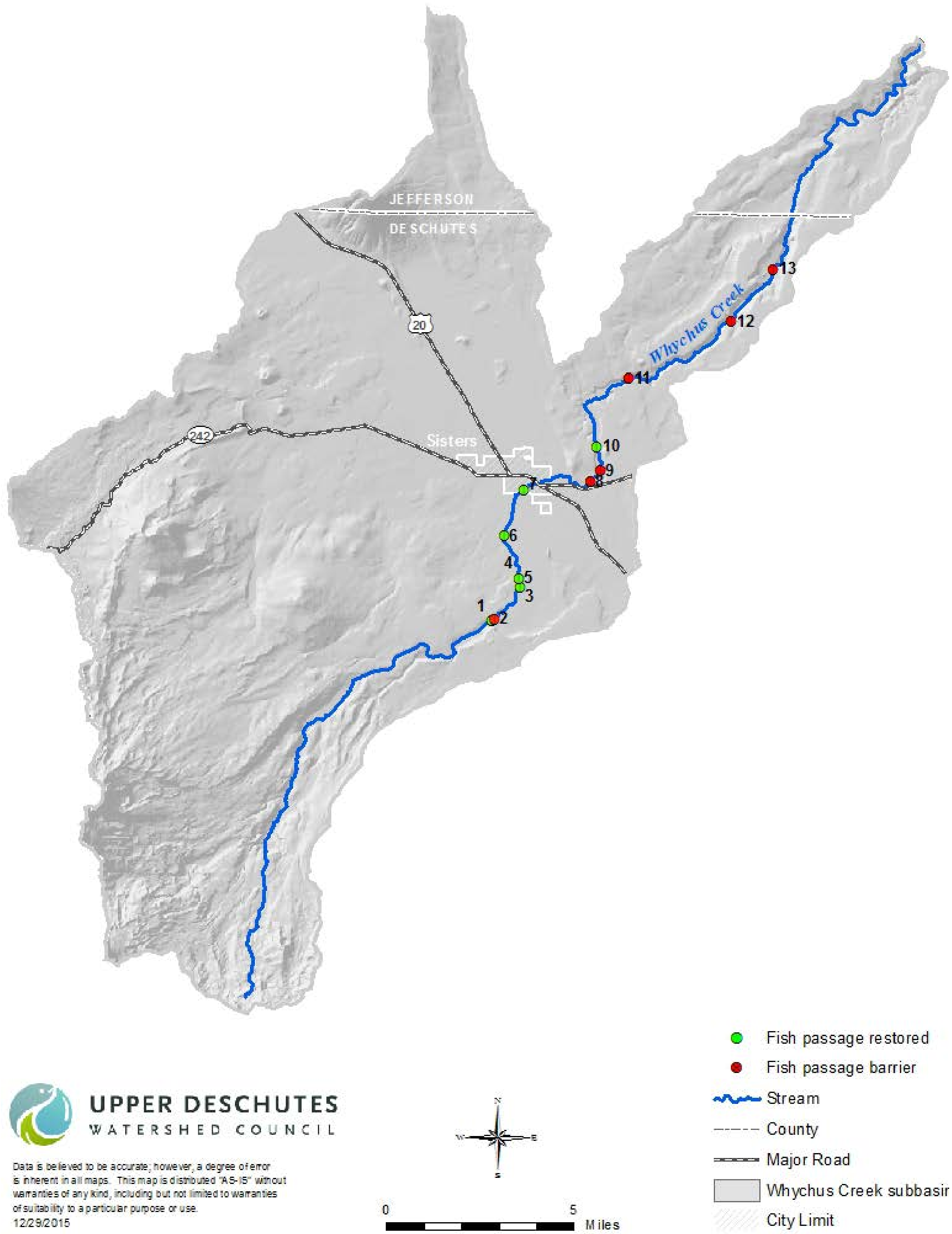

Figure 1.
The 2009 baseline inventory reported 12 of 13 irrigation diversions on Whychus Creek not to meet federal and state criteria for fish screens (NMFS 2008). From 2009 to 2014 cumulative flows diverted through unscreened diversions were reduced by 87%, to 22.45 cfs. Although the screen at Diversion no. 11 has not been verified to meet screening criteria, it is not considered to pose a risk of fish entrainment. Diversions No. 4 and 5 are located within feet of each other on opposite sides of the creek, thus only one of the two points is visible on the map.

Discussion

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

At the close of 2014, the cumulative unscreened diversion rate had been reduced by 87%, from 192.89 in 2009 to 22.45 cfs in 2014. Potential for fish entrainment in irrigation diversions has thus been dramatically reduced, owing greatly to the progressive practices of TSID management and to collaboration by water rights holders. UDWC and restoration partners will continue to engage with water rights holders and landowners to eliminate risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

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Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, and 2011-2014

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Abstract

This project has characterized the aquatic macroinvertebrate community in Whychus Creek and assessed changes following multiple restoration projects conducted across several years. Benthic macroinvertebrates were sampled in 2005, 2009, and 2011-2014 at 10-13 reaches along Whychus Creek, from RM 30.25 to RM 0.5. Xerces staff trained local volunteers in ORDEQ standardized sampling techniques for wadeable streams, and teams collected macroinvertebrate samples from riffles at pre-determined sites. Similarity among replicate site samples taken by Xerces staff for quality assurance indicates that trained volunteers implement the protocol successfully, and operator error has been minimized as much as is possible with different sets of volunteers doing the sampling in each year.

Biotic assessment techniques, i.e. the PREDATOR predictive model and the invertebrate-based index of biotic integrity established by the OR DEQ (ORDEQ IBI) have not shown dramatic improvements in biological conditions in Whychus Creek, apart from a trend towards increased IBI scores among upstream reach sampling sites. An IBI created for the Grande Ronde watershed, which may be more applicable to the Whychus Creek watershed, was used in 2014; this model assigned a better biological condition across all sites, but no dramatic changes were seen between years. However, multivariate analysis shows a marked shift in macroinvertebrate community composition between 2005/2009 and later years, and the many of the taxa that influenced between-year dissimilarities the most are OR DEQ indicators for temperature and percent fine sediment (FSS). Several taxa that are indicators for warm temperatures and/or high FSS had greater abundance in the 2005 assemblages, and some indicators for cool temperatures and low FSS were more abundant discriminator taxa in the 2011-2014 assemblages. Investigation of the individual metrics that comprise the ORDEQ revealed increases in sensitive taxa, and while site assemblages continue to have greater top-taxon abundance than would be expected in an unimpaired site, the identity of the top taxon among sites has changed, with the top taxa in earlier years tending towards more tolerant groups, and those in later years being more sensitive. Tolerant and sediment-tolerant taxa have shown a pattern of increase followed by decrease, which may reflect a response to the disturbance of restoration activities followed by stabilization and improved stream conditions. Analysis of functional feeding groups, which can have varying utility and was done for the first time this year, proved less informative.

Several aspects of the macroinvertebrate community show changes towards taxa with lower FSS and temperature optima, including the entire macroinvertebrate community at sampling sites, as well as the increaser/decreaser taxa and missing and replacement taxa identified though the PREDATOR model. The mean temperature optima of the macroinvertebrate community among all downstream and midstream sites has decreased significantly across time, and in 2005, both missing and decreaser taxa had a lower mean temperature optima than replacement and increaser taxa, while in 2011-2014, replacement and increaser taxa had lower mean temperature optima. The mean FSS optima of the

macroinvertebrate community at all sampling reaches has decreased significantly over time, and the mean FSS optima is lower among replacement and increaser taxa in almost every sampling year.

Multiple years of effectiveness monitoring along Whychus Creek have found changes in the benthic macroinvertebrate community. This time span should be sufficient for new colonists to find the stream and establish populations, if habitat conditions are adequate, but colonist diversity is dictated by the community composition in surrounding catchments and their connectivity with Whychus. Overall, the Whychus Creek benthic macroinvertebrate community saw the greatest changes from 2005 to 2009, but the community has continued to change in a direction that strongly indicates a response to lower temperature and fine sediment levels.

Project Background and Summary

Biomonitoring

Biomonitoring evaluates the biological health of a water body by examining its biotic communities, such as 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 a critical part of the food web in both aquatic habitats and the surrounding riparian system (Gibson, 1993; Sabo & Power, 2002; Baxter et al., 2005). They exhibit a range of responses to human-induced stressors, and changes in different groups can reflect the effects of temperature, substrate, habitat complexity, and flow. The phenology 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. Their short generation time can enable 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, and sampling and identification are relatively straightforward, standardized, and cost-effective.

Biomonitoring can be done to determine baseline biological conditions, investigate impacts of a disturbance or pollutant, and assess changes following restoration projects. Benthic macroinvertebrates are key biological indicators, as community composition can change over time in response to reach- and catchment-scale land management practices and habitat restoration activities (Albertson *et al*., 2011). Post-restoration biomonitoring can help document outcomes, justify expenditures, and guide future restoration projects (Palmer & Bernhardt, 2006). However, the amount of post-restoration biomonitoring that is done in the United States is very small compared to the ongoing level of restoration activity (Bernhardt et al., 2005). Data from existing projects often indicates varied and even conflicting outcomes, as many environmental factors interact at larger spatial scales to structure macroinvertebrate communities, the biological responses of different taxa may be non-linear, and the recolonization process in restored river segments is still poorly understood. The long-term monitoring that has been done in Whychus Creek can thus contribute important information to knowledge of post-restoration effectiveness monitoring.

Whychus Creek experienced significant degradation in the past from surrounding land use practices, including dewatering, 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 in 2005, 2009, and 2011-2014 to collect benthic macroinvertebrate samples along Whychus Creek 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; prior to this, the creek 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 done. Sampling in 2012-2014 occurred after the stream at Camp Polk Meadow was re-meandered; monitoring and maintenance of this site is still ongoing.

Biotic Assessment Techniques

Several analytical methods have been developed in an attempt to standardize assessments of stream biological communities. These approaches are tailored for different states and regions, but can be placed into two broad categories: multimetric indices and predictive models. In Oregon, these models have been developed primarily for the western regions of the state to assess assemblages collected from gravel and cobble substrates in riffle habitats.

Multimetric Indices

Multimetric indices rate a combination of community attributes (metrics) that have been found to respond predictably to human-induced stressors (Barbour et al., 1996; Karr & Chu, 1999). The raw values of individual metrics are converted to a standardized score (generally ranging from 1-5 or 1-10, depending on the model), then these standardized metric scores are summed to generate a single numerical index of biological integrity (IBI) value that reflects the biological condition of a site. A standard 10-metric IBI was 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). The Oregon IBI is comprised of six positive metrics (i.e. attributes that receive a higher standardized IBI score for a higher raw value) and four negative metrics (i.e., attributes that receive a lower standardized IBI score for a higher raw value). The positive metrics are expected to increase with improved biological conditions, while the negative metrics are expected to decrease.

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 developed for the state (Hubler, 2008 and pers. comm.). In 2014, we became aware that a second and more regionalized IBI had been developed in the past specifically for use in the Grande Ronde watershed (Shannon Hubler, pers. comm.) in northeastern Oregon. The two models use essentially the same metrics, with the exception that the more widely used ORDEQ IBI rates the percent dominance of the top taxon while the Grande Ronde IBI rates the percent dominance of the top three taxa, but the raw values are scored according to a different scale (see Table 1 for a comparison of metrics and scoring). Because the Grande Ronde watershed may be more similar to the Whychus Creek watershed than the coastal and western watersheds used to develop the statewide IBI, both IBIs were applied to the existing Whychus Creek dataset. Although neither may be ideal, it should be noted that regardless of the final summed score generated, regular assessment of stream condition using a macroinvertebrate IBI enables detection of changes and trends in community composition, especially among sensitive taxa.

Table 1.
Comparison of general Oregon DEQ and Grande Ronde IBI metrics and scoring. Both use taxa identified to genus/species.

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

**For ORDEQ IBI, dominance of the top (most abundant) taxon is assessed; for the Grande Ronde IBI, abundance of the top 3 taxa is assessed.*
***MHBI = Modified Hilsenhoff Biotic Index, reflecting tolerance to organic pollution/enrichment; individual taxa MHBI may range from 1 (low tolerance) to 10 (high tolerance).*

PREDATOR Predictive Model

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; MWCF) and the Western Cordillera and Columbia Plateau (Klamath Mountain, Cascades, East Cascades, Blue Mountains, and Columbia Plateau ecoregions; WCCP). The model calculates the ratio of taxa observed at a sampling site to taxa expected at that site if no impairment existed (O over E), based on community data collected previously at a large number of reference streams. The model incorporates environmental gradients such as elevation, slope, and longitude when selecting reference streams. An O/E value of less than one indicates loss of taxa, while values greater than one indicate taxa enrichment, potentially in response to pollution or nutrient loading. In addition to calculating sample O/E scores, the model also generates a probability of capture for individual taxa at each sampling site, allowing specific taxa loss and replacement to be investigated. Additional model output includes a sensitivity index, calculated as the # of sites at which a taxon was observed / # of sites at which the taxon was expected, enabling evaluation of taxa that are 'increasers' and 'decreasers' in the stream. Under the WCCP model, biological condition is assigned to a site based on the following O/E scores: ≤0.78 = poor (most disturbed); 0.79 – 0.92 = fair (moderately disturbed); 0.93 – 1.23 = good (least disturbed); >1.23 = enriched.

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

Methods

Stream Sampling Sites

Thirteen stream reaches were sampled along Whychus Creek in 2014; of these, 11 had been sampled in multiple previous years, while two were new sampling sites (see Table 2 for all sampling site locations and years). Duplicate samples were taken at two sites for quality control purposes. While some sites were added or removed from the slate since 2005 based on access, changes in land use, or re-assessment of their importance, overall the sampling sites are distributed broadly into downstream (RM 0.5- 10.75; DS), mid-stream (RM 18-19.5; MS), and upstream reaches (RM 23.5 - 30.25; US).

Table 2.
Whychus Creek sampling sites, 2005-2014.

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, 2011-2014
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, 2011-2014
WC00650	RM 6.5, u/s Rd 6360 crossing	44.396799, -121. 404268	2013
WC00875[d,f]	RM 8.75, Rimrock Ranch d/s	44.391278, -121.406182	2011-2014
WC00900	RM 9, Rimrock Ranch	44.384198, -121.407892	2005, 2009, 2011-2014
WC00925[e]	RM 9.25, Rimrock Ranch	44.380815, -121.408592	2013
WC00950	RM 9.5, Rimrock Ranch u/s	44.371534, -121.415865	2011-2012, 2014
WC01025	RM 10.25	44.364587, -121.421706	2014
WC01075	RM 10.75	44.361288, -121.427525	2014
WC01800	RM 18	44.328342, -121.494534	2005
WC01825	RM 18.25, d/s end DBLT property	44.32781, -121.495406	2009, 2011-2014
WC01850	RM 18.5, DBLT property	44.326601, -121.500229	2009, 2011-2014
WC01900	RM 19, DBLT property	44.321523, -121.507461	2005, 2009, 2011-2014
WC01950	RM 19.5, d/s Camp Polk Bridge, DBLT	44.318741, -121.514961	2009, 2011-2014
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, 2011-2014
WC02600[c,f]	RM 26, 4606 Rd. footbridge	44.2730592, -121.555297	2005, 2009, 2011-2014
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

Superscripts 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; [f] 2014 duplicate

Volunteer Training

Volunteer training and stream sampling was done in the same index period across all years, i.e. the Saturday falling within August 17-20. Watershed council and Xerces 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 technique was demonstrated and each item on the data sheet was explained. A handout with detailed step-by-step sampling instructions was provided, as well as field guides to Northwest stream macroinvertebrates (Adams *et al.*, 2003), although volunteers were not expected to identify organisms. The group was then divided into teams of two 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; plastic squirt bottle; plastic spoons; forceps; thermometer; tape measure; 10-gallon plastic bucket; hand lens; 1-liter Nalgene sample jars half-filled with 80% ethanol; datasheets; jar labels; clipboard; and detailed instructions for finding site(s). Volunteers also had the cell phone numbers of Xerces and UDWC staff, and staff members moved among the sites throughout the day to be available to help and answer questions. All samples were returned to Xerces staff at the park at the end of the day, and staff immediately inspected the jars to be sure each was appropriately labeled and preserved.

Sampling Procedures

Benthic macroinvertebrate samples were collected from riffle habitats according to standardized protocols developed by the Oregon Department of Environmental Quality (ORDEQ) for Oregon's wadeable streams (see OWEB, 2003). Sampling reach lengths were calculated as 40 times the average wetted width of the stream at the desired sampling point, within a minimum of 500 feet and a maximum of 1000 feet. In 2005 and 2009, volunteers calculated the wetted width and measured reach lengths, but since 2009, watershed council staff performed these calculations and flagged the upstream and downstream extent of each reach a few days prior to sampling.

Eight randomly selected riffles within each stream reach were sampled. Each sample consisted of eight net sets, each collected from a 1 ft x 1 ft 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 rubbed and rinsed into the net to dislodge and collect any clinging organisms and set aside. The substrate was then disturbed thoroughly using a boot heel or hand to a depth of 6-10 cm for ~60 seconds. The eight individual net samples at each site were pooled into a bucket; large debris was rinsed and removed, and any vertebrates such as fish were noted and carefully replaced in the stream. Sample material was concentrated by being poured through a 500 µm sieve lined with a flexible 500 µm Nitex membrane; the membrane was then lifted out and the concentrated sample was carefully scooped and rinsed into the sample jar.

Samples containing excessive amounts of sand and gravel were elutriated, which separates the lighter invertebrates from heavier mineral material and allows them to be jarred separately to avoid crushing or grinding them. Elutriation was done 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 rinses, the organic material was placed in sample jars separate from the mineral material, but all sample material from each site was retained for subsequent examination in the lab so that heavier-bodied organisms (i.e., snails, case-making caddisflies) were not lost. Jars were filled no more than halfway with sample to ensure preservation, and the ethanol was replaced within 48 hours to maintain the 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.

Macroinvertebrate Identification

Samples were identified by the laboratory of Mike Cole from 2009-2014 (via Cole Ecological, Inc. in 2013-2014, and ABR, Inc. Environmental Research & Services in 2009-2112); those collected in 2005 were identified by Robert Wisseman (Aquatic Biology Associates, 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 2014, only the sample from RM 6 contained fewer than 500 organisms. From 2005-2013, the target count of 500 was not attained for only from one to three samples from within a given year.

All organisms picked from the samples were identified to the level of taxonomic resolution currently used by the OR DEQ (genus and species). If a specimen was too immature for key taxonomic characters to be fully developed or visible, identification was done only to the family level.

Data analysis

Invertebrate Index of Biotic Integrity (IBI)

Biological condition of each sampling site was assessed using both the Level 3 IBI developed by OR DEQ, as well as the more regional Grande Ronde IBI (Table 1), and the correlation between the two sets of scores was determined. All sites from all sampling years were assessed using both IBI models, and the changes in scores of mean raw values as well as mean scaled values among upstream, mid-reach, and downstream sites were assessed across time. Two-tailed t-tests were conducted in Excel to determine significance of differences between years.

PREDATOR model

Invertebrate community data were analyzed using the PREDATOR predictive model for the Western Cordillera + Columbia Plateau (WCCP; Hubler, 2008). Observed over expected (O/E) scores associated with a probability of capture (P_c) > 0.5 were used (i.e. the model considers only invertebrates with over 50% likelihood of being collected at reference sites).

PREDATOR scores are generated from a site habitat file and a sample data file which are uploaded to 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 provides a measure of biological condition; a probability matrix indicating missing taxa that are expected to occur at each site but are absent, and replacement taxa that are not expected to occur at the site but are present in the sample; and a sensitivity index 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.

Temperature and Sediment Optima

Differences in macroinvertebrate community composition may be driven by stream temperature and the amount of fine sediment in the substrate. OR DEQ developed a dataset containing optima values for seasonal maximum temperature and percent fine sediments (i.e. the temperature and %FSS under which a taxon can maximize its abundance). The temperature and sediment optima of increaser vs. decreaser taxa and of missing vs. replacement taxa were examined to assess differences and diagnose whether the macroinvertebrate community was responding to changes in temperature and/or sediment conditions. The weighted mean temperature and sediment optima of the macroinvertebrate community at each sampling site was also assessed and examined across sites and years. We further noted the presence of taxa considered by OR DEQ to be indicators (i.e. taxa with the strongest responses to environmental gradients) of cool or warm temperatures and high or low fine sediment conditions among the missing, replacement, increaser, and decreaser taxa (Hubler et al., 2009; see Table 3 for a list of OR DEQ indicator taxa).

Table 3.
OR DEQ indicator taxa for temperature and fine sediment stressors. Values in parentheses indicate temperature (^{o}C) or sediment (% fine sediment) optima values for each taxon.

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

Taxon	Temperature	Fine sediment
Paraleptophlebia	---	High (11)
Tanypodinae	---	High (12)
Ostracoda	---	High (17)
Hydroptila	---	High (17)
Lymnaeidae	---	High (18)
Cheumatopsyche	---	High (20)
Sphaeriidae	---	High (21)
Coenagrionidae	---	High (25)

Taxonomic and Ecological Trait Analysis

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

To incorporate ecological traits into bioassessment of Whychus Creek, a trophic guild (functional feeding group, FFG) was assigned to each taxon in the dataset from 2005-2104. Designation of a taxon as a predator, scraper, shredder, collector-filterer, or collector-gatherer was done according to Merritt et al. (2008); the richness, relative diversity, and relative abundances of taxa in different FFGs was determined for each year's dataset, and changes in distributions among downstream, mid-reach, and upstream sites was examined.

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

Community Similarity Analysis

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

Results and Discussion

Invertebrate Index of Biotic Integrity

ORDEQ IBI

The OWEB Level 3 stream IBI 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 (most abundant) 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). Table 4 shows the IBI score for each sampling site from 2005-2014.

No site in any year received an IBI score that indicated severely disturbed conditions. Moderately disturbed conditions were seen at 33% of sites in 2005, 0 sites in 2009 and 2012, 29% of sites in 2011, 17% of sites in 2013, and 31% of sites in 2014. The majority of sites in all years received an assessment of slightly disturbed: 56% in 2005, all sites in 2009, 57% in 2011, 82% in 2012, 75% in 2013, and 69% in 2014. Only a few sites in a few years received scores indicating minimal/no disturbance, including 11% in 2005, 14% in 2011, 18% in 2012, and 8% in 2013. However, no sustained increase in IBI scores has occurred.

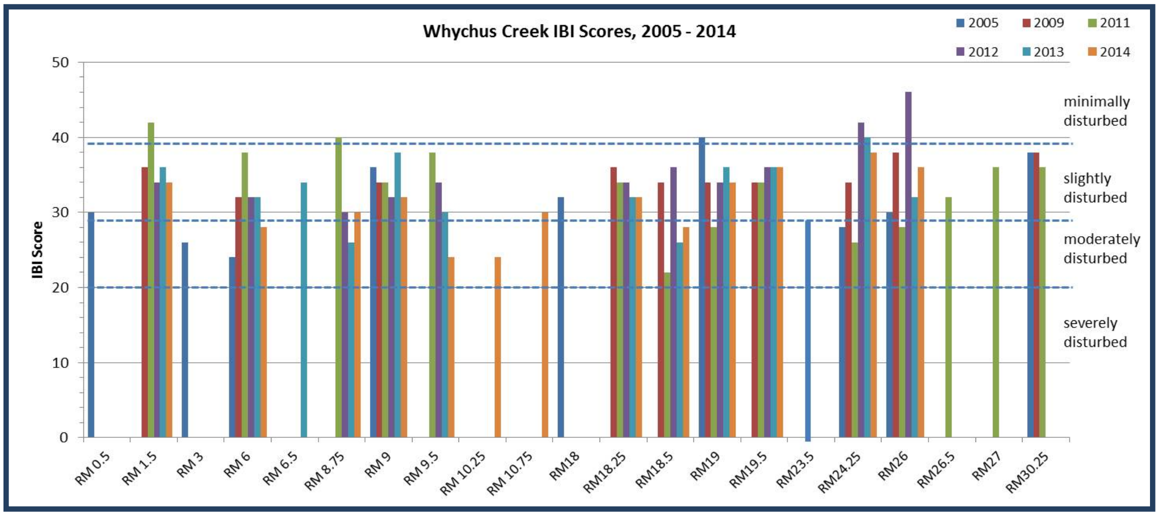
Table 4
ORDEQIBI scores for all sites samples between 2005 and 2014. Colors indicate minimal (blue), slight (green), or moderate (lavender) disturbance.

Site	2005	2009	2011	2012	2013	2014
WC00050	30	---	---	---	---	---
WC00150	---	38	44	34	36	34
WC00300	26	---	---	---	---	---
WC00600	24	32	38	32	32	28
WC00650	---	---	---	---	34	---
WC00875	---	---	40	30	26	30
WC00900	36	34	34	32	38	32
WC00950*	---	---	38	34	30	24
WC01025	---	---	---	---	---	24
WC01075	---	---	---	---	---	30
WC01800	32	---	---	---	---	---
WC01825	---	36	34	34	32	32
WC01850	---	34	22	36	26	28
WC01900	40	34	28	34	36	34
WC01950	---	34	34	36	36	36
WC02325	28	---	---	---	---	---
WC02425	28	34	26	42	40	38
WC02600	30	38	28	46	32	36
WC02650	---	---	32	---	---	---
WC02700	---	---	36	---	---	---
WC03025	38	38	36	---	---	---

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

When examining changes in mean IBI scores across different stream reaches across time, sampling sites in the downstream and upstream reaches showed the greatest changes, although differences overall have not been dramatic (Figure 1). Mean IBI scores among the downstream reaches were significantly higher in 2011 ($p \leq 0.05$) compared to previous years, but the mean IBI score in 2012-2014 was significantly lower than in 2011, although from 2012-2014 the differences in mean IBI scores were not significant. A trend for increasing scores can be seen among the upstream sites, with mean IBI scores increasing through 2012. Upstream mean IBI scores were lower in both 2013 and 2014 than in 2012, but the differences were not significant. Less change was seen in the mean IBI scores among all mid-stream sites; all years remained within the “slight disturbance” scoring range, and differences between mean scores in any years were not significant.

a. Individual site scores across time



b. Changes in IBI scores across time among different stream reaches. Letter pairs indicate significant difference between mean values (p<0.05). Downstream = RM 0.5-10.75; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

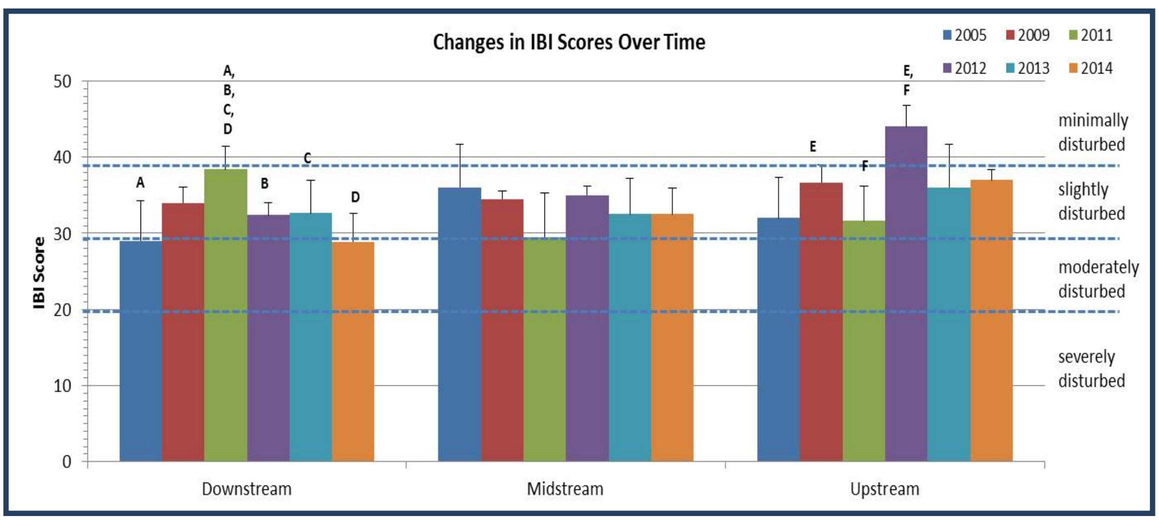


Figure 1.
ORDEQ IBI scores for Whychus Creek sampling sites.

Grande Ronde IBI

In past years we have had concerns that the ORDEQ IBI, which is better calibrated for the Coast range watersheds in which it was developed, may not accurately score sampling sites in this eastern river in an arid landscape. In 2014, we became aware that an additional, more regional IBI had also been developed by ORDEQ for the Grande Ronde watershed (GR IBI; Shannon Hubler, pers. comm.). The metrics used in the GR IBI are the same as those used in the ORDEQ IBI, with the exception that instead of scoring the dominance of the single most abundant taxon (ORDEQ IBI), the dominance of the three most abundant taxa is sored (GR IBI). However, the raw score ranges and corresponding scaled scores are different between the two IBI tools, and the GR IBI summed scores reflect only three levels of biological condition (minimal impairment, >25; moderate impairment, 15-25; and severe impairment,

<15), in contrast to the four categories in the ORDEQ IBI (minimal, slight, moderate, and severe impairment; see Table 2). Although the GR IBI is not widely used, it was applied to the all sites sampled on Whychus Creek to assess any differences in biological condition that might be revealed, and to determine how well the ORDEQ and GR IBI scores correlated with each other.

Differences in scoring between the two IBI models resulted in a much greater proportion of sites receiving a score indicating minimal disturbance (70% in 2005, 100% in 2009 and 2012, 71% in 2011, 92% in 2013, and 77% in 2014), with the remainder scoring as moderately disturbed (Table 5; Figure 2a).

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

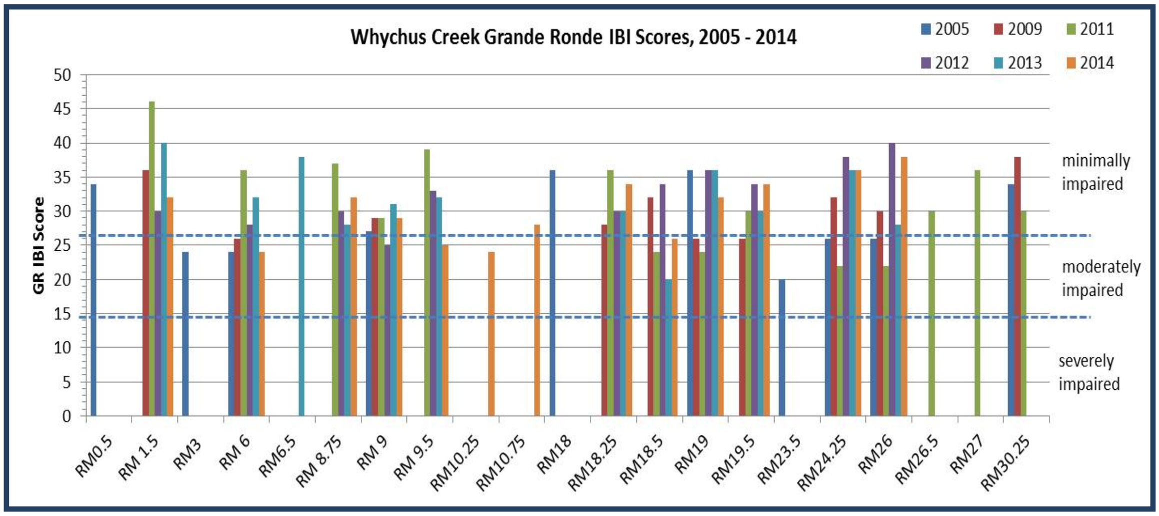
Site	2005	2009	2011	2012	2013	2014
WC00050	34	---	---	---	---	---
WC00150	---	36	46	30	40	32
WC00300	24	---	---	---	---	---
WC00600	24	26	36	28	32	24
WC00650	---	---	---	---	38	---
WC00875	---	---	37	30	28	32
WC00900	27	29	29	25	31	29
WC00950*	---	---	39	33	32	25
WC01025	---	---	---	---	---	24
WC01075	---	---	---	---	---	28
WC01800	36	---	---	---	---	---
WC01825	---	28	36	30	30	34
WC01850	---	32	24	34	20	26
WC01900	36	26	24	36	36	32
WC01950	---	26	30	34	30	34
WC02325	20	---	---	---	---	---
WC02425	26	32	22	38	36	36
WC02600	26	30	22	40	28	38
WC02650	---	---	30	---	---	---
WC02700	---	---	36	---	---	---
WC03025	34	38	30	---	---	---

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

Mean IBI scores among the downstream, midstream, and upstream sampling segments all indicated minimal impairment, in contrast to the mean ORDEQ IBI scores among reach segments which indicated a mixture of slight, moderate, and severe impairment. When the mean GR IBI scores were examined among the three different reach segments across time, the overall patterns of change observed were similar to those seen for the ORDEQ IBI, though a trend towards increased mean scores was slightly more pronounced among upstream and downstream sites (Figure 2b). This similarity in pattern was borne out by the strong correlation between the ORDEQ and GR IBI scores when plotted against each other on a site-by-site basis (R^2 = 0.6298; Figure 3). Among the downstream reaches, a similar trend of increased mean IBI score was seen, although the scores in 2012-2014 were in this case significantly lower than the mean IBI score in 2012. The upward trend in mean GR IBI score among the upstream reaches was also similar to the pattern seen using the ORDEQ IBI, with scores rising significantly through 2012 (note that the difference between the mean upstream IBI scores in 2005 and 2014 was close to significance, at p = 0.07), and showing no significant decrease afterwards. Under the GR IBI model, changes in mean IBI scores among the midstream reaches were more substantial, with the mean score in 2009 significantly lower than in both 2005 and 2012, but still not indicative of any sustained increase or decrease.

Scoring scales for the Grande Ronde IBI may be better adapted for conditions in Whychus Creek, as the setting of this watershed is likely more similar to that of Whychus Creek than any coastal watershed, and this model indicates better biotic conditions overall. Both IBI models further suggest a sustained trend towards improving biological conditions in the upstream reaches of the creek, less substantial change with no strong trends in the midstream reaches, and greater variation in the downstream reach sites, which may indicate conditions in greater flux or a greater mixture of stressors.

a. Individual site scores for GR IBI across time



b. Changes in GR IBI scores across time among different stream reaches. Letter pairs indicate significant difference between mean values (p<0.05). Downstream = RM 0.5-10.75; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

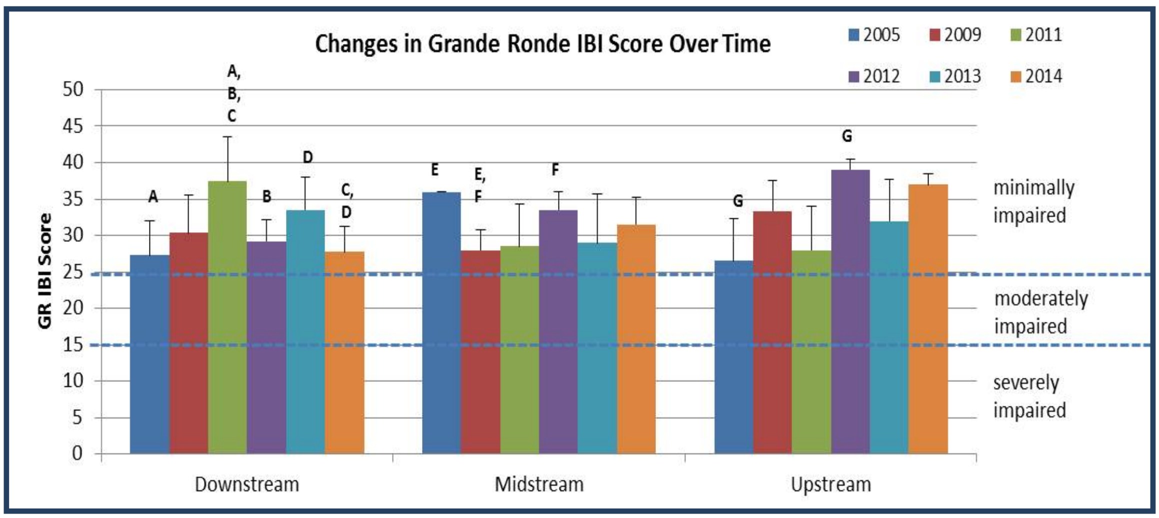


Figure 2.
Grande Ronde IBI scores for Whychus Creek sampling sites.

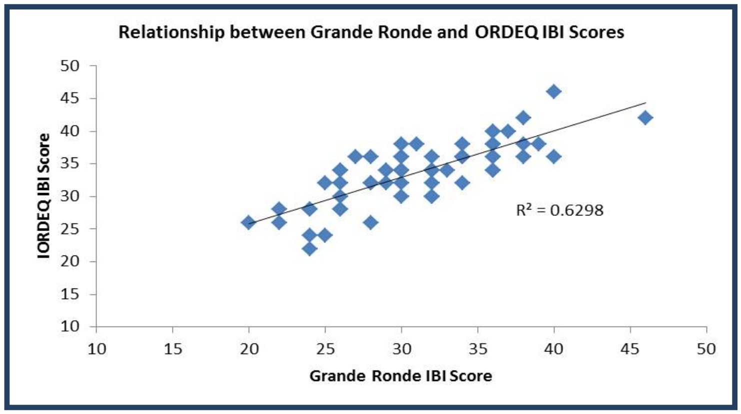


Figure 3.
Correlation between ORDEQ and Grande Ronde IBI scores for Whychus Creek

Predator

Due to concerns about a continuing trend of overall low O/E scores among the sampling sites, we consulted with ORDEQ staff to be sure that the model was being used correctly, and shared the raw site data to allow them to conduct a PREDATOR analysis independently. Their O/E results were identical or very similar, with a few individual exceptions that were likely due to differences in the overall community composition resulting from the taxa subsampling step (this is a required part of the model in which the full taxonomic data set for each site is randomly subsampled from 500 down to a total of 300 per site using a piece of software that accompanies the PREDATOR model). However, in the course of discussions with DEQ staff and examination of the datasets, it became apparent that in 2005, the values in the habitat file were incorrect (specifically for mean annual precipitation and temperature). The model was run again on that year's data with the corrected values, resulting in changes to the 2005 PREDATOR scores (note that the IBI was not affected by this).

Site O/E

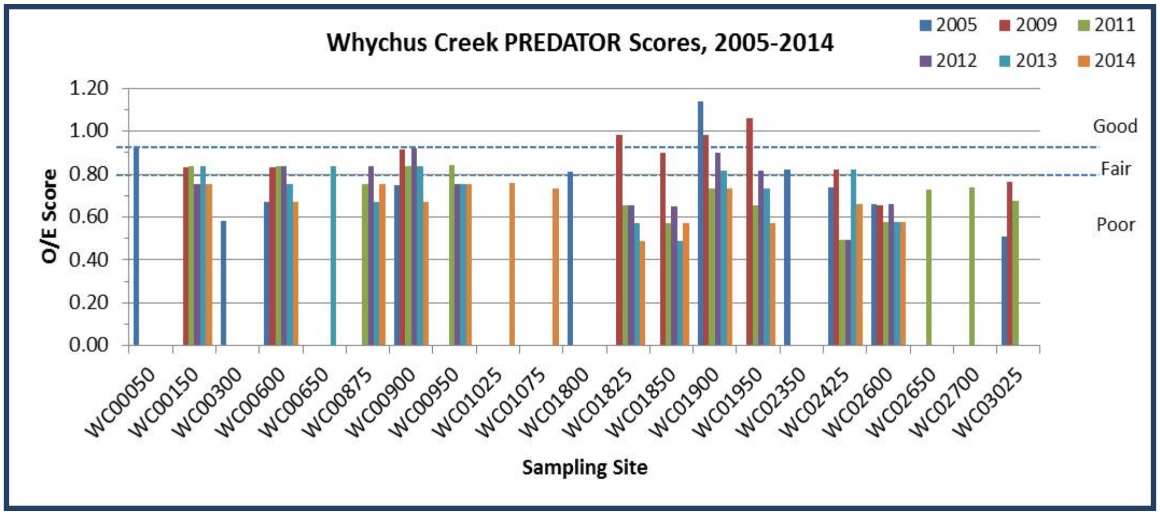
PREDATOR scores have in general indicated both worse biological conditions and a smaller degree of change over time than expected (Table 6, Figure 4a). Among sites sampled across multiple years, some have shown little change in O/E score (i.e. WC00150, WC00950), some have shown a downward trend (i.e. WC01825, WC01850), and some have fluctuated substantially (i.e. WC01900, WC02425). In contrast to what was observed for the IBI results, mid-reach scores have shown the greatest change over time, with mean O/E scores from 2011-2014 significantly lower than in 2005 and 2009 (Figure 4b). Mean O/E scores among downstream reach sites remained similar across all sampling years until 2014, when they were significantly lower. Greater variation was seen among mean O/E scores in the upstream reaches, but differences were not significant in any year.

Table 6.
PREDATOR O/E scores. Colors indicate biological conditions that are good (least disturbance, blue), fair (moderate disturbance, green), or poor (severe disturbance, lavender).

Site	2005	2009	2011	2012	2013	2014
WC00050	0.92	---	---	---	---	---
WC00150	---	0.83	0.84	0.75	0.84	0.75
WC00300	0.58	---	---	---	---	---
WC00600	0.67	0.83	0.84	0.84	0.75	0.67
WC00650	---	---	---	---	0.84	---
WC00875	---	---	0.75	0.84	0.67	0.75
WC00900	0.75	0.92	0.84	0.92	0.84	0.67
WC00950*	---	---	0.84	0.76	0.75	0.76
WC01025	---	---	---	---	---	0.76
WC01075	---	---	---	---	---	0.73
WC01800	0.81	---	---	---	---	---
WC01825	---	0.98	0.65	0.65	0.57	0.49
WC01850	---	0.90	0.57	0.65	0.49	0.57
WC01900	1.14	0.98	0.73	0.90	0.82	0.73
WC01950	---	1.06	0.65	0.82	0.73	0.57
WC02350	0.82	---	---	---	---	---
WC02425	0.74	0.82	0.49	0.49	0.82	0.66
WC02600	0.66	0.66	0.58	0.66	0.58	0.58
WC02650	---	---	0.73	---	---	---
WC02700	---	---	0.74	---	---	---
WC03025	0.51	0.76	0.68	---	---	---

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

a. Individual site O/E scores across time



b. Changes in PREDATOR O/E scores across time among different stream reaches. Letter pairs indicate significant difference between mean values ($p<0.05$). Downstream = RM 0.5-10.75; midstream = RM 18.0-19.5; upstream = RM 23.5 – 30.25.

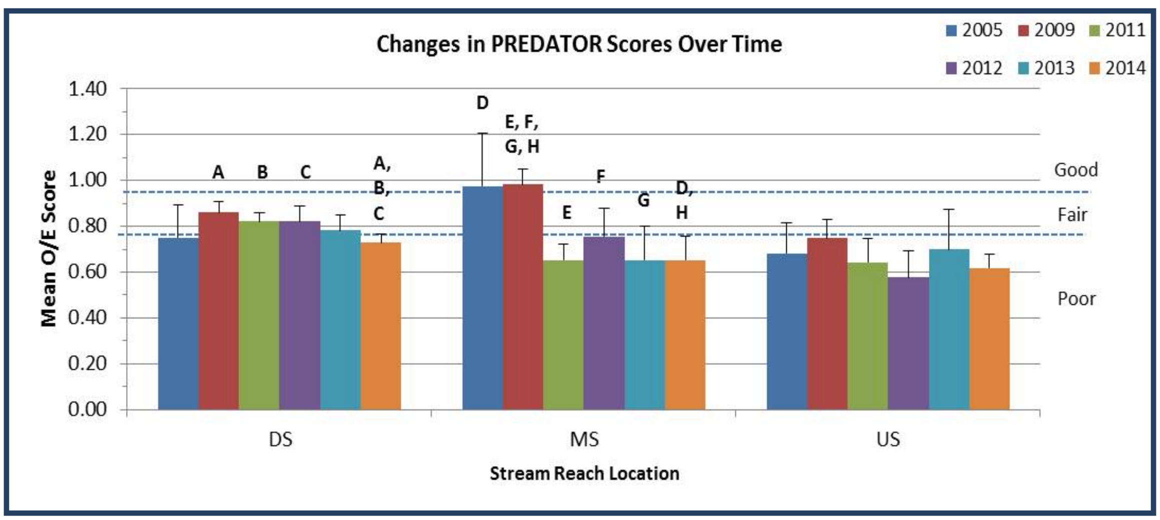


Figure 4.
PREDATOR O/E Scores for Whychus Creek sampling sites

Increaser and Decreaser Taxa

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

Interestingly, both increaser and decreaser taxa included several that have been identified by ORDEQ as indicator taxa for temperature and/or sediment, based on their optima values for temperature and %FSS (see Table 7 for summary of indicator taxa). Results show an increasing number of cool temperature and low FSS indicator taxa among the increasers; for example, in 2005 there were no cool temperature indicator taxa among the increasers whereas there were three cool temperature indicators among the decreaser taxa, while every subsequent year had at least one cool temperature indicator among the increaser taxa, and from 2011-2014 several warm temperature indicator taxa were among the decreasers. Similarly, while every year has at least one low sediment indicator taxon among both the increasers and decreasers, the number of low sediment indicators among the increasers increased, while the number of high sediment indicator taxa rose among the decreasers and dropped among the increaser taxa across years. These results suggest a change in macroinvertebrate community composition in response to improving stream conditions.

Table 7.

Number of indicator taxa for temperature and fine sediments present within PREDATOR increaser and decreaser taxa groups.

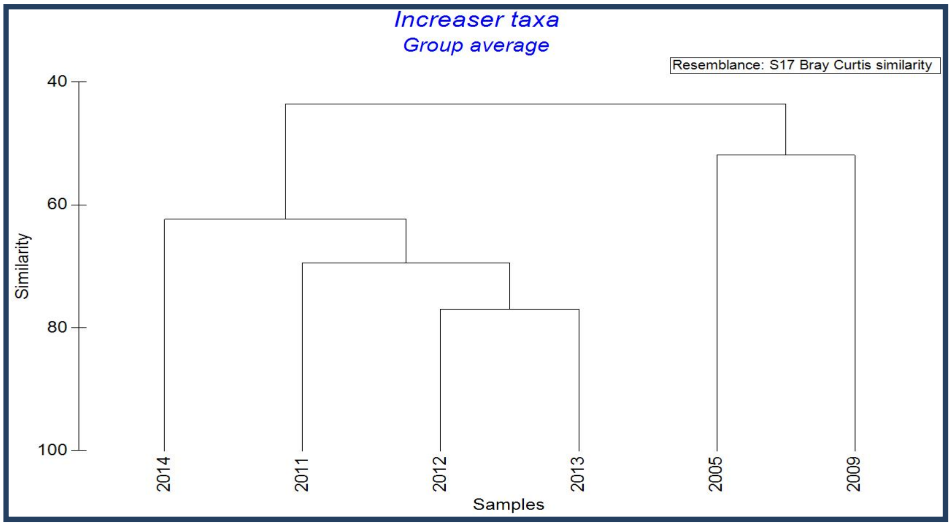
	2005		2009		2011		2012		2013		2014	
	INC	DEC	INC	DEC	INC	DEC	INC	DEC	INC	DEC	INC	DEC
Cool indicator	0	3	1	3	3	4	2	4	3	5	1	7
Warm indicator	0	0	0	0	0	4	0	4	1	3	1	4
Low FSS indicator	1	1	1	1	3	1	3	1	3	1	3	6
High FSS indicator	0	0	1	2	1	5	1	4	0	4	0	1

A few taxa occurred as increasers in each sampling year: *Rhithrogena* (flatheaded mayfly genus with low tolerance for organic pollution; DEQ low sediment indicator), Capniidae (a sensitive stonefly family with low tolerance for organic pollution), *Acentrella* (a genus of small minnow mayfly), and *Brachycentrus* (a genus of humpless casemaker caddisfly with low tolerance for organic pollution). A greater number of taxa were seen as decreasers in all sampling years, including *Lepidostoma* (a genus of casemaker caddisflies with low tolerance for organic pollution), *Malenka* (a genus of small brown stonefly with low tolerance to organic pollution), *Micrasema* (a genus of humpless casemaker caddisfly with low tolerance for organic pollution; DEQ cool temperature indicator), Pisidiidae (a family of sediment- and pollution-tolerant fingernail clams; DEQ high sediment indicator), *Heterlimnius* (a genus of riffle beetle), *Drunella doddsi* (a sensitive species of spiny crawler mayfly with low tolerance for organic pollution; DEQ cool temperature and low sediment indicator), *Ironodes* (a genus of flatheaded mayfly; DEQ cool temperature indicator), *Neophylax* (a genus of stonecase caddisfly), and *Maruina* (a moth fly genus).

A CLUSTER analysis of a Bray-Curtis similarity matrix of presence/absence abundance of increaser and decreaser taxa showed differences in these communities across time (Figure 5). For both the increaser and decreaser taxa, the community was most similar between the 2005 and 2009 samples, and among the 2011-2014 samples, although a greater average percent similarity was seen among decreaser groups compared to increaser groups.

These results show that despite the lack of agreement between PREDATOR and IBI condition scores, the macroinvertebrate community is clearly changing across time in response to changing sediment and temperature conditions. The presence of increasing numbers of warm temperature and high fine sediment indicators among the decreaser taxa group since 2005 suggests improving temperature and sediment conditions in the stream overall. These differences were examined further by assessing the mean temperature and sediment optima values of the increaser and decreaser communities in each year (see "Temperature and Sediment Optima", below).

a. Increaser taxa communities



b. Decreaser taxa communities

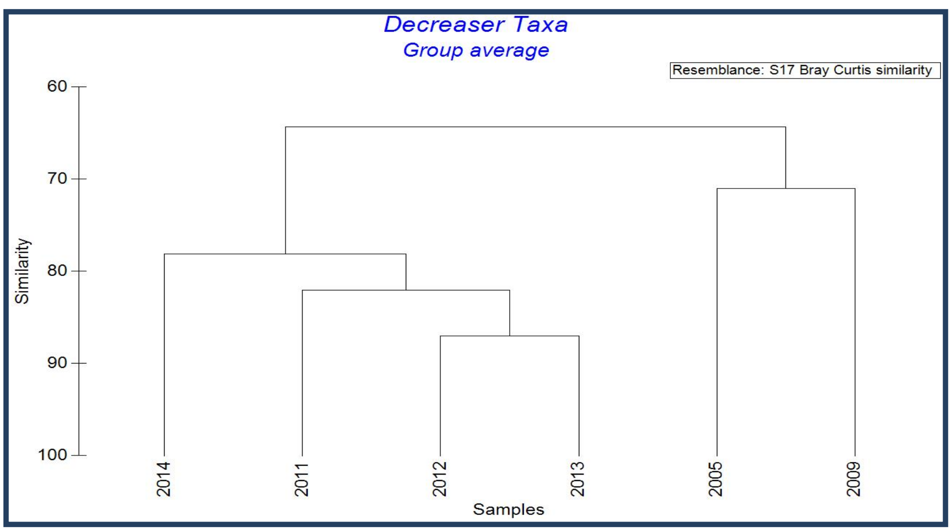


Figure 5.
Similarity among increaser and decreaser taxa groups in different sampling years

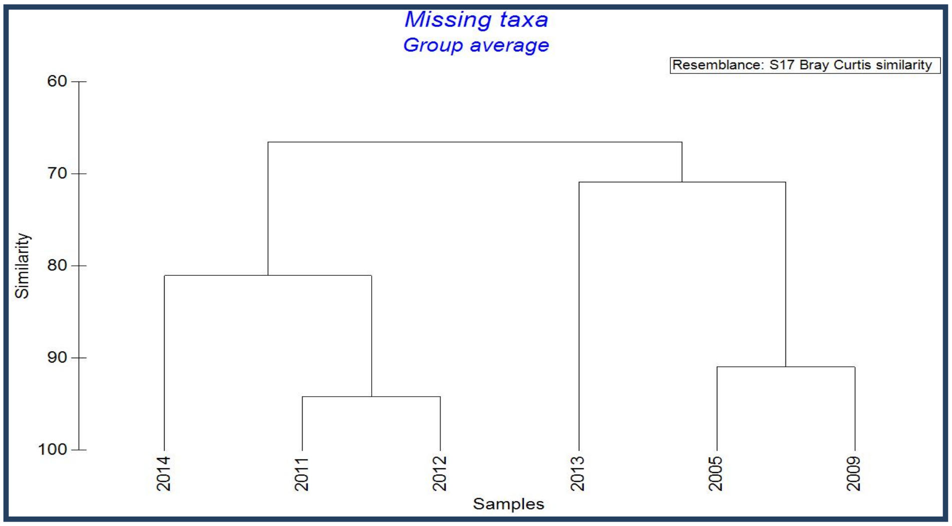
Missing and Replacement Taxa

A third output from the PREDATOR model identifies taxa that were expected at a site but not collected (missing taxa), and taxa that were not expected at a site but were collected in the sample (replacement taxa). Missing and replacement taxa are generated for each individual sampling site. There was a great deal of similarity among both missing and replacement taxa from 2005 to 2014, when taxa absent from ≥7 sites are considered.

Taxa identified as missing from ≥7 sites in each sampling year included *Malenka* (a genus of small brown stonefly with low tolerance to organic pollution), Pisidiidae (a family of sediment- and pollution-tolerant fingernail clams; DEQ high sediment indicator), Tanypodinae (a common non-biting midge group; DEQ high sediment indicator), and *Calineuria* (a moderately sensitive perlid stonefly genus). *Malenka* and Pisidiidae were also among the decreaser taxa in each year. Taxa identified as replacements at ≥7 sites in each sampling year included Diamesinae (a non-biting midge group), *Serratella (*a commonly-collected genus of spiny crawler mayfly with low tolerance to organic pollution), *Rhithrogena* (a genus of flatheaded mayfly with low tolerance for organic pollution; DEQ low sediment indicator), *Acentrella* (a small minnow mayfly genus), *Narpus* (a common, moderately tolerant riffle beetle genus), and *Atherix* (a common, tolerant watersnipe fly genus). *Rhithrogena* and *Acentrella* were also among taxa identified as increasers in each year.

A CLUSTER analysis of a Bray-Curtis similarity matrix of presence/absence missing and replacement taxa showed differences in these communities across time (Figure 6). However, in contrast to the increaser and decreaser communities, which showed similar between-year clustering patterns for both groups, similarity patterns among missing and replacement taxa differed across years. The community of missing taxa was most similar between 2005 and 2009 (90.9%), but this pair was more closely related to the missing community in 2013 than in the next contiguous sampling years. The replacement taxa communities were most similar between 2005 and 2014 (82.8%), while the contiguous sampling years of 2011-2013 clustered separately (74.7% similar). However, there was a great deal of overall similarity within the missing community across all years, with only 28% of all missing taxa being unique (i.e. occurring as missing at ≥7 sites in only a single year), and only 15% of all replacement taxa being unique (occurring as replacement at ≥7 sites in only a single year).

a. Missing taxa communities



b. Replacement taxa communities

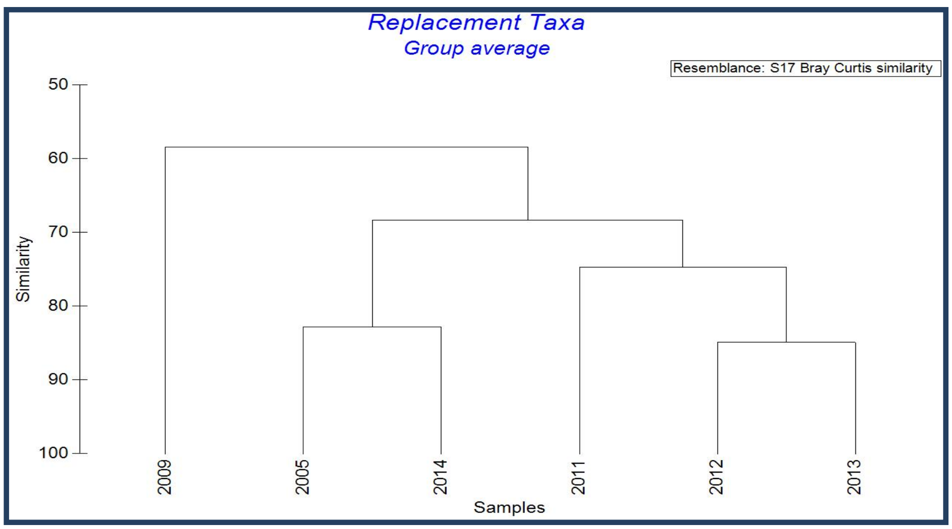


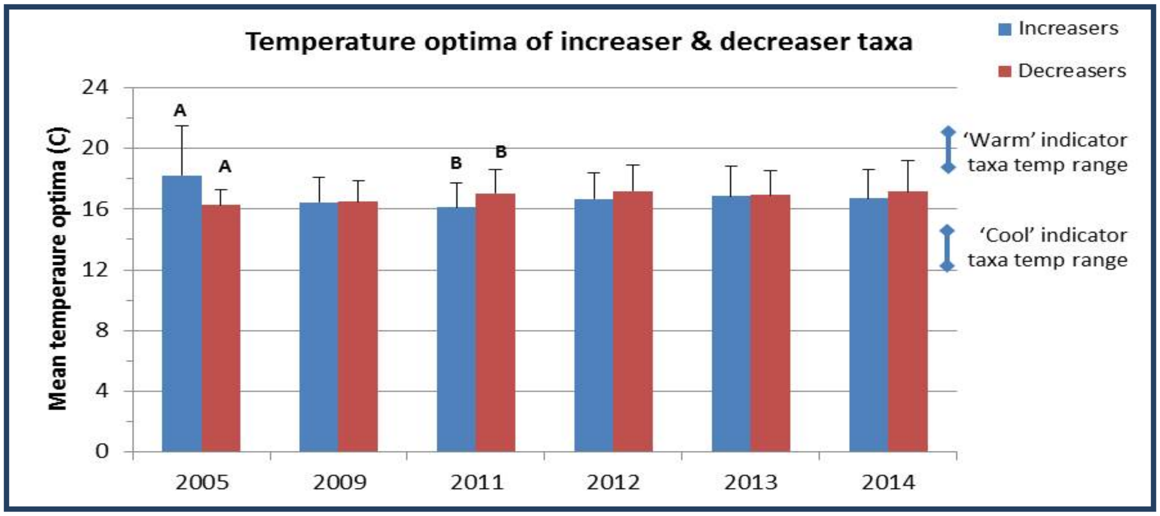
Figure 6.
Similarity among missing and replacement taxa groups in different sampling years

Temperature and Sediment Optima

Differences in sediment and temperature optima among indicator taxa in increaser vs. decreaser communities across time, combined with changes in the missing and replacement community composition, led us to investigate whether sediment and/or temperature was a driving force in observed community changes. We therefore compared the mean temperature and sediment optima among increaser vs decreaser taxa; missing vs. replacement taxa; and the complete macroinvertebrate community collected at each site in each sampling year. In all cases, significant differences were observed for both temperature and sediment, although responses to lower sediment appear to be a stronger driver.

Increaser taxa show a clear trend for lower temperature and sediment optima values compared to decreaser taxa, although differences were not significant in every year (Figure 7). The greatest difference is seen in sediment optima, which are higher in every year among the decreaser tax. This difference was significant in 2012 and 2014, and close to significant (p = 0.06) in 2011. While the differences in temperature optima between increaser and decreaser communities were less pronounced, there is an interesting trend. In 2005, the mean temperature optima of the increaser taxa was significantly higher than that of the decreaser taxa; in 2009, the mean temperature optima of the two communities was almost identical (16.45^{o}C vs. 16.51^{o}C); and in every subsequent year except 2013 (when the optima of the two groups was again almost identical), the mean temperature optima of the increaser community was lower than that of the decreaser community, with the difference being significant in 2011. Many organisms can function best within a narrow range of temperatures, such that even small shifts can have a large impact on a community. Thus, although the differences between temperature optima of the increaser and decreaser communities are not great, they suggest a pattern of response to lower stream temperatures.

a. Mean temperature optima of increaser and decreaser taxa across time. Letter pairs indicate a significant difference between mean values (p<0.05). The range of temperatures spanned by ORDEQ indicator taxa is shown for reference.



b. Mean percent fine sediment optima of increaser and decreaser taxa across time. Letter pairs indicate a significant difference between mean values (p<0.05). The range of %FSS values spanned by ORDEQ indicator taxa is shown for reference.

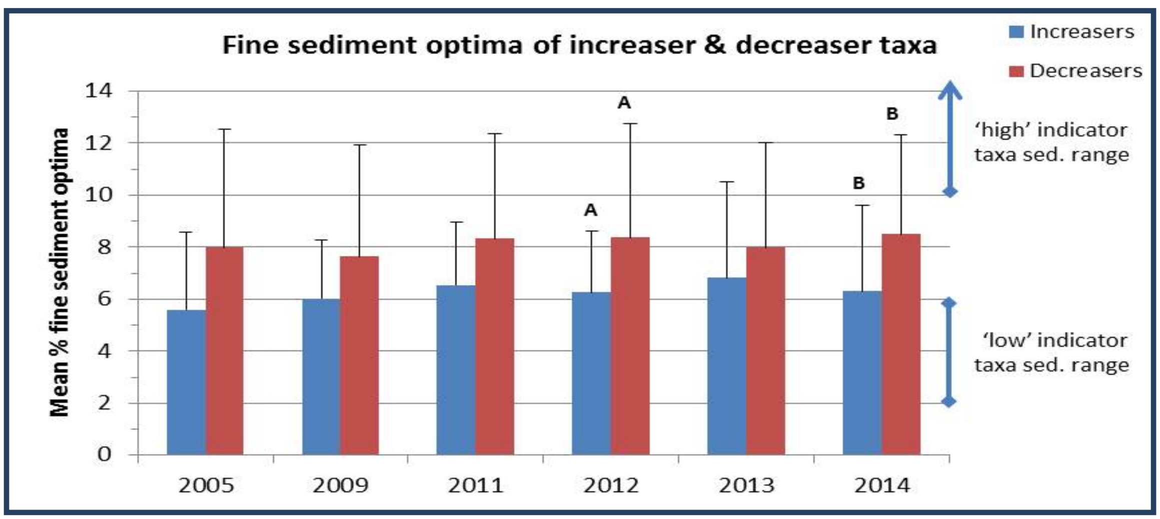
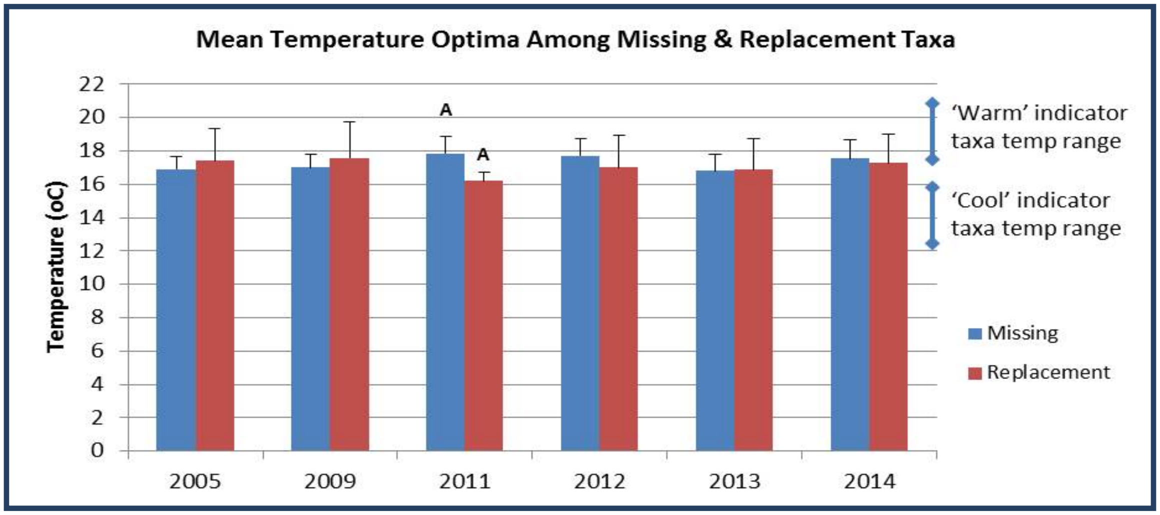


Figure 7.
Differences in temperature and fine sediment tolerances among increaser and decreaser taxa

The pattern is similar although less pronounced when comparing the optima of missing and replacement communities across sampling years (Figure 8). In earlier years, mean temperature optima of replacement taxa are lower than missing taxa, while in later years replacement taxa have a lower mean temperature optima compared to replacement, although the difference is significant only in 2011. The same strong trend for lower mean fine sediment optima seen for increaser taxa occurs again among replacement taxa, and the difference in mean fine sediment optima is significant between missing and replacement communities in every sampling year except for 2009 (but note the difference between the means in 2009 is close to significant, at p=0.056).

a. Mean temperature optima of missing and replacement taxa across time. Letter pairs indicate a significant difference between mean values (p<0.05). The range of temperatures spanned by ORDEQ indicator taxa is shown for reference.



b. Mean fine sediment optima of missing and replacement taxa across time. Letter pairs indicate a significant difference between mean values (p<0.05). The range of %FSS spanned by ORDEQ indicator taxa is shown for reference.

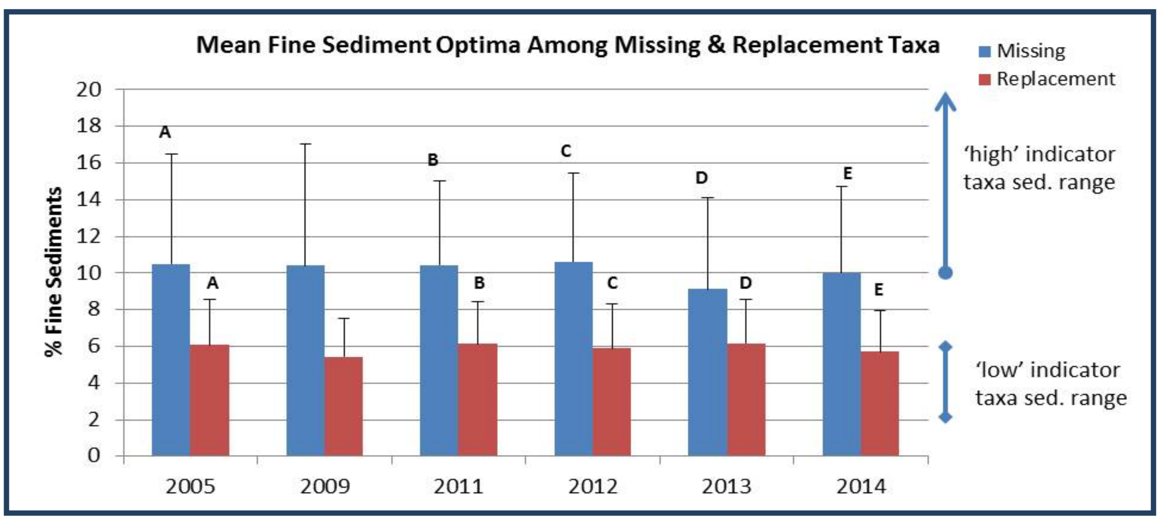
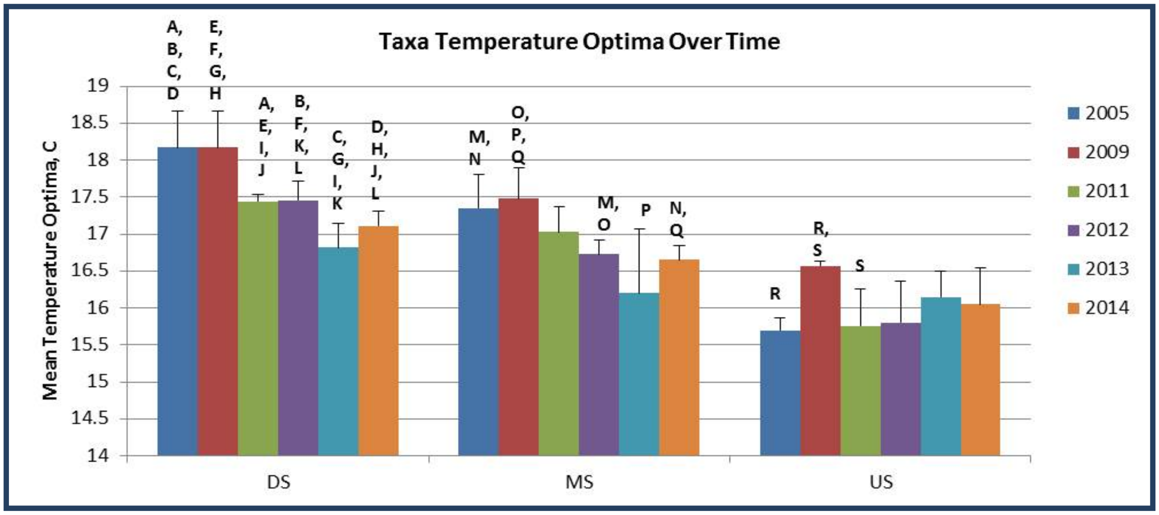


Figure 8.
Differences in temperature and fine sediment tolerances among missing and replacement taxa

Finally, we compared the weighted means of temperature and sediment optima of the communities collected at each site in all sampling years, and examined changes across time within different stream segments (Figure 9). 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. As would be expected, the macroinvertebrate assemblages collected at all upstream sampling sites have the lowest overall mean temperature optima and the assemblage at downstream sampling sites has the highest mean temperature optima. This agrees with the expected changes in stream conditions and biota 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. Within this overall pattern, however, changes among the mean temperature optima can be detected across time among all stream reaches. These changes are most evident within the downstream sampling sites, where a stepwise and significant decrease in mean temperature optima is visible from 2005/2009 to 2011/2012 and 2013/2014 (Figure 9a). A similar downward trend is also clearly evident among mid-stream sampling sites, with temperature optima among the community present in 2012-2014 significantly lower than those of the communities in 2005 and 2009. Mean temperature optima in this section of the stream may be stabilizing, as there were no significant differences among the mean temperature optima for the assemblages in 2012, 2013, and 2014, although results suggest a downward trend may still be occurring. Mean temperature optima among the upstream site communities, which are the lowest of all the stream reaches, have shown less change throughout the years; a significant increase seen in 2009 dropped again in 2011 to about the same mean value as was seen in 2005, and although a slight increase in mean temperature optima has been seen in subsequent years, it has not been significant in any.

A consistent decrease in mean fine sediment optima has also occurred in multiple years among downstream, mid-stream, and upstream sampling sites (Figure 9b). Overall, the mean sediment optima of the macroinvertebrate assemblage present among all sampling sites in 2012-2014 has been significantly lower than that of the assemblages present at the same reaches in 2005-2009.

a. Changes in weighted means of macroinvertebrate community temperature optima.



b. Changes in weighted means of macroinvertebrate community % fine sediment optima.

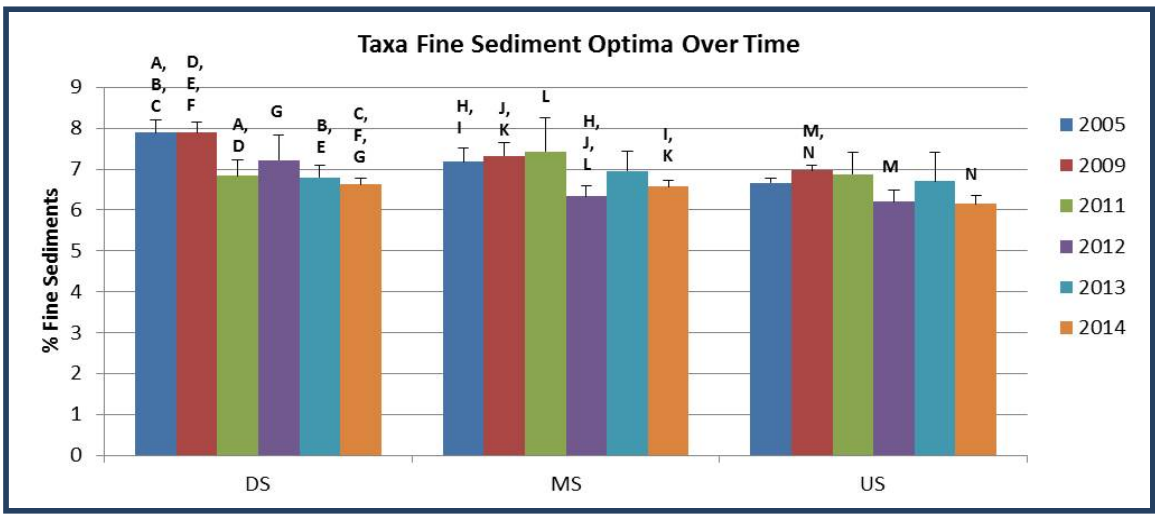


Figure 9.
Changes in temperature and fine sediment tolerances among macroinvertebrate assemblages collected along Whychus Creek. Letter pairs indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Taxonomic and Ecological Trait Analysis

The data above illustrate that while changes in PREDATOR and IBI scores have not been dramatic, the macroinvertebrate community assemblage has clearly changed over time. Based on comparisons of temperature and fine sediment optima values among missing, replacement, increaser, and decreaser taxa, as well as the total community present at each site, the community appears to be responding to lower stream temperatures and reduced fine sediments. We therefore decided to examine changes in the raw values of each of the metrics that comprise the ORDEQ to see if this would be more informative than the total scaled IBI score for each site. We also examined changes in the diversity and abundance of taxa in different functional feeding groups at each sampling site, as this can also be influenced by different habitat conditions and stressors.

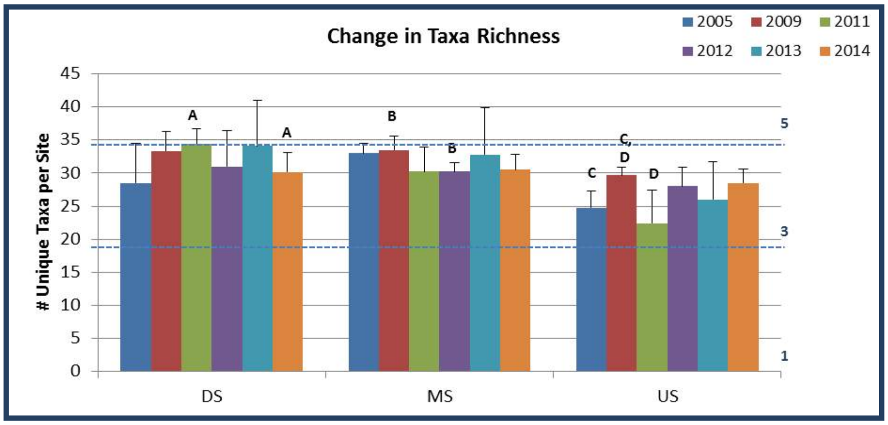
Diversity measures

In general, a healthy stream is expected to have higher diversity (number of unique taxa), as there are more intact habitats and microhabitats to support a variety of sensitive and tolerant organisms with multiple feeding, habitat, and reproductive behaviors. For this reason, “taxa richness” is a positive metric in the ORDEQ IBI, and the PREDATOR O/E model measures taxon loss. However, it should be noted that some healthy freshwater habitats such as small cold headwater streams and nutrient- or mineral-poor waters may be naturally less diverse, while recurring low or intermediate levels of disturbance may result in an increase in taxa richness (Townsend et al., 1977). Also, the fixed-count subsampling routinely used when identifying macroinvertebrate samples may not account for every species or genus present in the total sample. Standardized subsampling targets can range from 100-500 organisms, and this study uses the 500 organism target to detect help the majority of unique taxa.

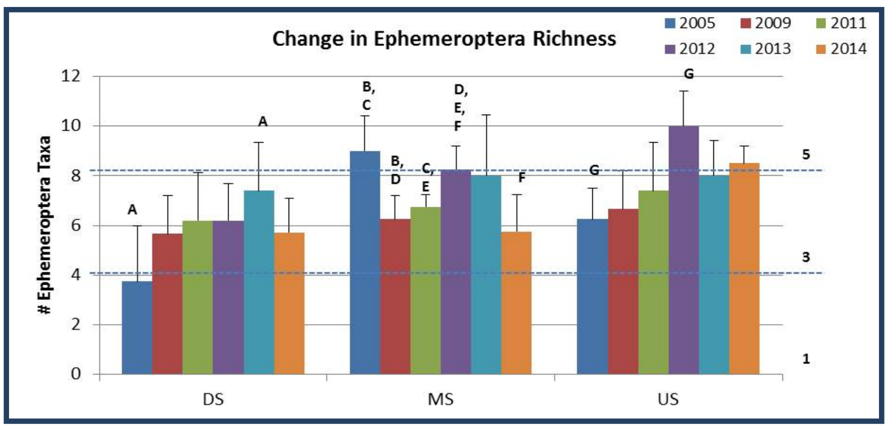
The diversity of individual taxa or taxa groups that are considered to be particularly sensitive indicators can also be assessed. The ORDEQ IBI scores the richness of Ephemeroptera (mayflies), Trichoptera (caddisflies), and Plecoptera (stoneflies) as three separate metrics; these are often combined into a single EPT richness metric in other IBIs, and we have examined the total richness, relative diversity, and relative abundance of EPT in Whychus Creek samples in past years. Although tolerance values differ at the individual species level, these groups as a whole are considered to be the most sensitive to increases in stream temperature, sedimentation, and organic pollution (Barbour et al., 1996).

Changes in taxa richness have not been pronounced (Figure 10a). Mean richness has oscillated between years among all sampling reaches. Examination of the diversity of sensitive EPT taxa reveals a different picture, with an overall increase among the three groups individually and as a whole. A pattern of increasing diversity was seen most clearly for Ephemeroptera (Figure 10b); diversity of this group increased significantly among downstream and upstream sampling reaches across years, with greater oscillation in diversity seen among mid-stream sites. Plecoptera richness showed a similar though not significant pattern (Figure 10c), with an overall increase in diversity across the years among downstream and upstream sampling sites, and greater variation among mid-stream sampling sites. Variation in mean richness was greatest among Trichoptera, with large oscillations. The trend among downstream reaches appears to be towards decreasing Trichoptera richness, while diversity upstream sites has increased (though not significantly) and diversity at mid-stream sites has remained roughly constant apart from a significant decrease in 2011 (Figure 10d).

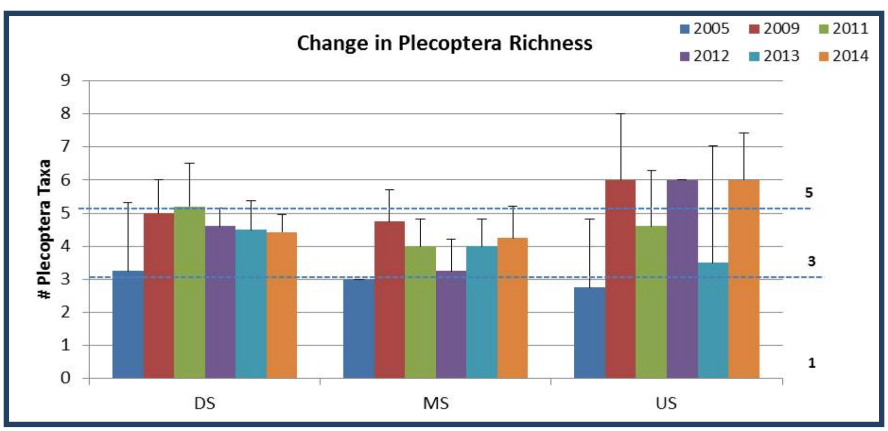
a. Changes in taxa richness



b. Changes in Ephemeroptera richness



c. Changes in Plecoptera richness



d. Changes in Tricoptera richness

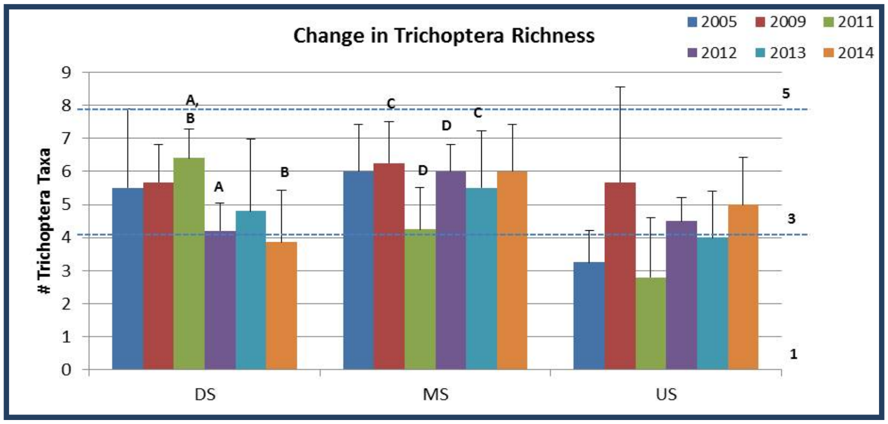
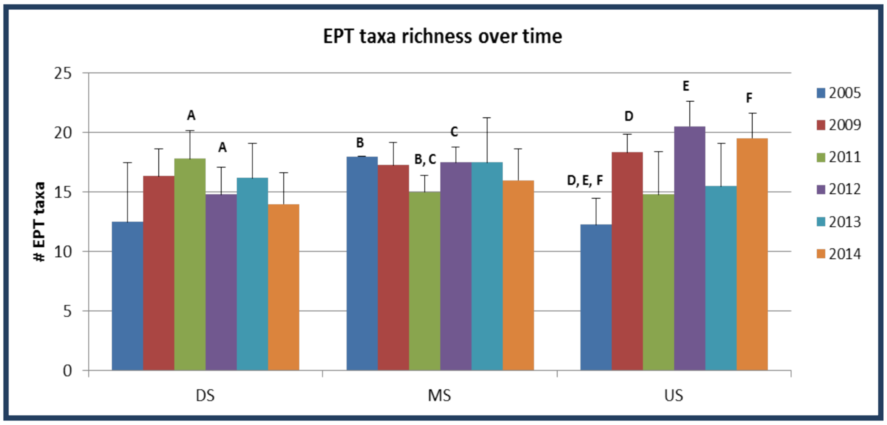


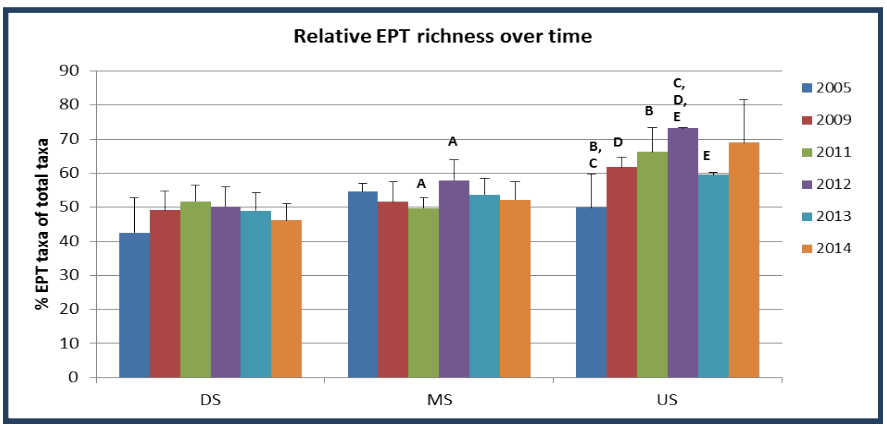
Figure 10.
Changes in diversity metrics among Whychus Creek sampling sites. Letter pairs indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25. Ranges for IBI scoring (5, 3, 1) are indicated.

Examination of the EPT as a combined group showed a trend towards increased diversity among upstream and downstream sites, with mean number of taxa remaining more constant among mid-stream reaches (Figure 11a), and the proportion of total taxa comprised of EPT increased significantly among upstream sites (Figure 11b). The most significant changes were seen when the relative abundance of EPT individuals was examined, with significant increases in the proportion of EPT individuals among the total sample organisms at downstream, mid-stream, and upstream sampling sites. In 2013 the relative abundance of EPT taxa among mid-stream sites showed a significant decrease, but the proportion of EPT individuals was again significantly higher in 2014 (Figure 11c).

a. Changes in EPT richness



b. Changes in EPT relative diversity



c. Changes in EPT relative abundance

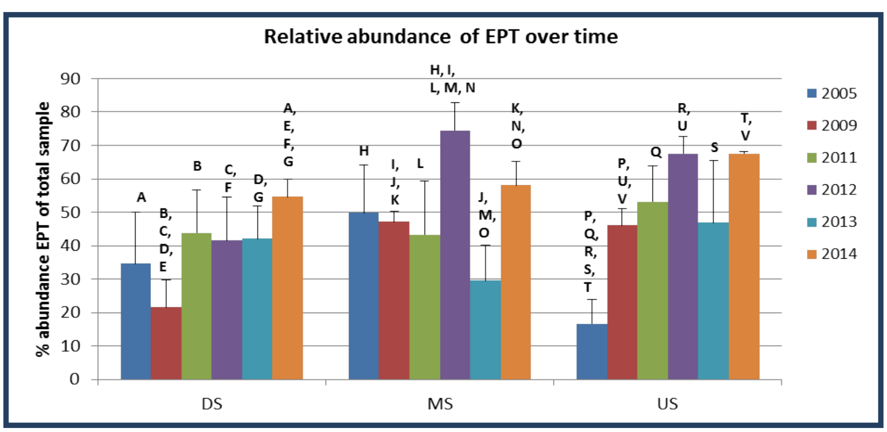


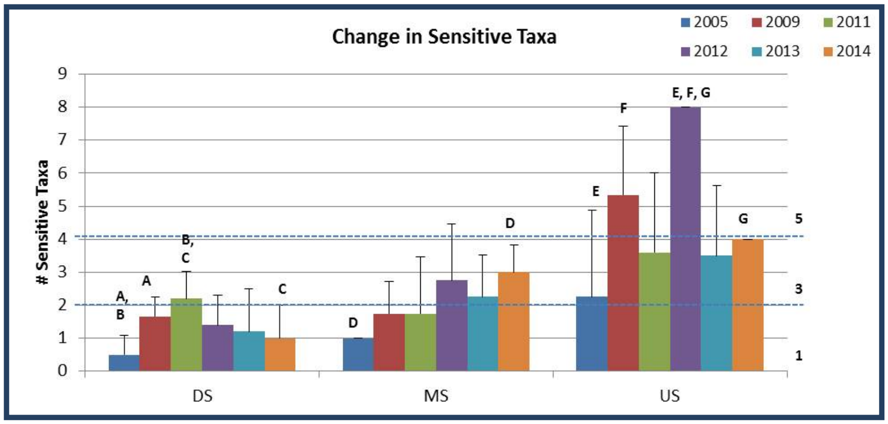
Figure 11.
Changes in the EPT assemblage among Whychus Creek sampling sites. Letter pairs indicate significant difference between mean values (P<0.05). Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Sensitivity measures

Metrics in the ORDEQ IBI that relate to sensitivity include number of sensitive taxa (i.e. groups sensitive to pollution and disturbance) and # sediment-sensitive taxa, which are very intolerant to inputs of fine sediment and likely to be absent from systems where sediments are a stressor. Upstream sites in general show the greatest number of sensitive taxa across time (Figure 12a). The number of taxa sensitive to disturbance has risen more steadily over the years among mid-stream reaches, and a more punctuated increase has occurred at upstream reach sites. The mean number of sensitive taxa among upstream sites increased overall from a low of 2.25 in 2005 to a high of 8 in 2012, but the mean number of sensitive taxa among upstream sites was lower in both 2011 and 2013, so it remains to be seen whether this punctuated increase will continue, especially as the mean number of sensitive taxa was again higher in 2014 than in 2013, though the difference was not significant. An initial significant increase in the number of sensitive taxa from 2005-2011 at downstream sites has changed to a downward trend in subsequent years, although the number was not significantly lower than 2011 until the current year. This may reflect disturbances at downstream sampling sites, but the overall number of sensitive taxa has consistently been very low at those sites (ranging from 0-3 among all downstream sampling sites in all years), so the magnitude of these differences overall is small.

The sustained pattern of increases in taxa with low sediment optima among increaser and replacement taxa and among the macroinvertebrate community collected at each site is not reflected in the changes in sediment-intolerant taxa (Figure 12b). The mean numbers of sediment-sensitive taxa have varied among sites and years, as is seen by the large standard deviations in the mean values, and the means have varied from year to year, showing both increase and decreases. However, the overall range of sediment-intolerant taxa numbers is again quite small, ranging from only 0-2 among all sites in all years, so this individual metric is limited in information content.

a. Changes in number of sensitive taxa



b. Changes in number of sediment-sensitive taxa

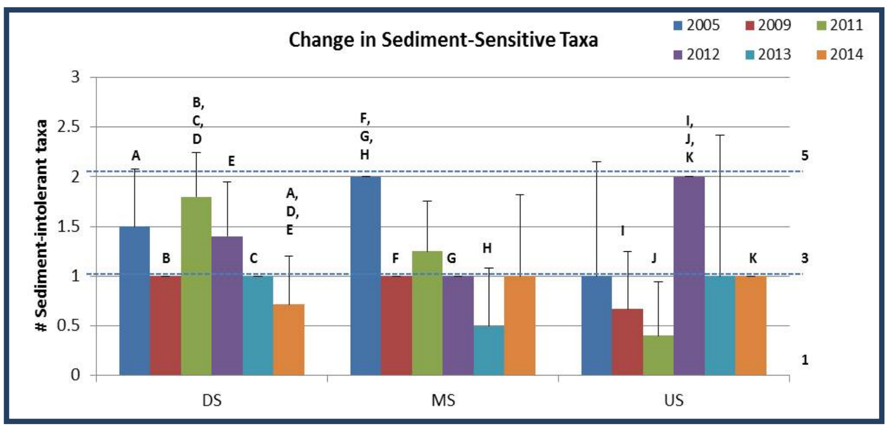


Figure 12.
Changes in sensitivity measures among Whychus Creek sampling sites. Letter pairs indicate significant difference between mean values (P<0.05). Ranges for IBI scoring (5, 3, 1) are indicated. Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Dominance

Dominance refers to the contribution of any individual taxon to the overall abundance of organisms in a sample. A healthy community is considered to support a diverse group of organisms and no single taxon, regardless of its traits (i.e. sensitive, tolerant, etc.), should overwhelm the community. Thus, an increase in taxon dominance is considered to reflect an increase in stream disturbance. In the ORDEQ IBI, a community whose top taxon has a relative abundance of <20% receives the highest score.

Figure 13 shows the changes in mean % dominance of the most abundant taxon across time at different stream segments. The dominance of the top taxon among upstream sites has decreased since 2005 from a mean of 56% (i.e. more than half of the total sample abundance comprised of a single taxon) to 28% in 2013 and 2014. In 2005 and 2009, upstream sampling sites were dominated by Orthocladiinae (a subfamily of non-biting midge common in cold, flowing waters), while in 2013 and 2014, they were dominated by *Baetis tricaudatus* (a common and somewhat tolerant small minnow mayfly species; DEQ low sediment indicator) and *Rhithrogena* (a flatheaded mayfly genus with low tolerance for organic pollution; DEQ low sediment indicator) as well as Orthocladiinae.

Mean % dominance among midstream sites decreased from 2005 through 2009 (35% and 19%, respectively), then increased through 2013 (48%) before falling again in 2014 (39%), although the difference in means between these two years was not significant. In 2005 and 2009, the dominant taxa among midstream sites included *Simulium* and *Baetis tricaudatus* as well as *Zapada* (a genus of small brown stonefly sensitive to organic pollution) and Oligochaeta (disturbance- and sediment-tolerant aquatic earthworms; DEQ high sediment indicator); in 2013 and 2014, dominant taxa at these reaches were *Simulium* and Orthocladiinae (2013), and *Baetis tricaudatus* (DEQ low sediment indicator; 2014).

Mean % dominance among downstream sites showed a U-shaped pattern, with a high of 33% in 2005 dropping to a low of 18% and then rising again through 2014, where the mean value was 36%. *Zaitzevia* (a tolerant riffle beetle genus; DEQ warm temperature and high sediment indicator) was the main dominant taxon at downstream sites in 2005, while in 2014 *Baetis tricaudatus* (DEQ low sediment indicator) was the dominant taxon.

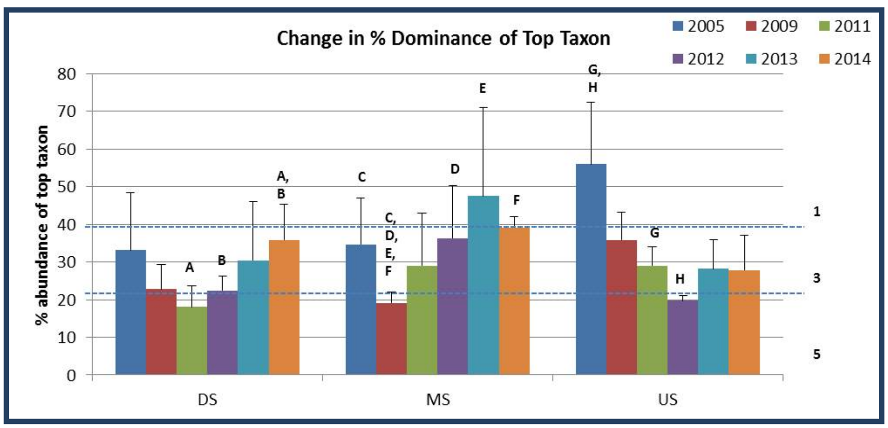


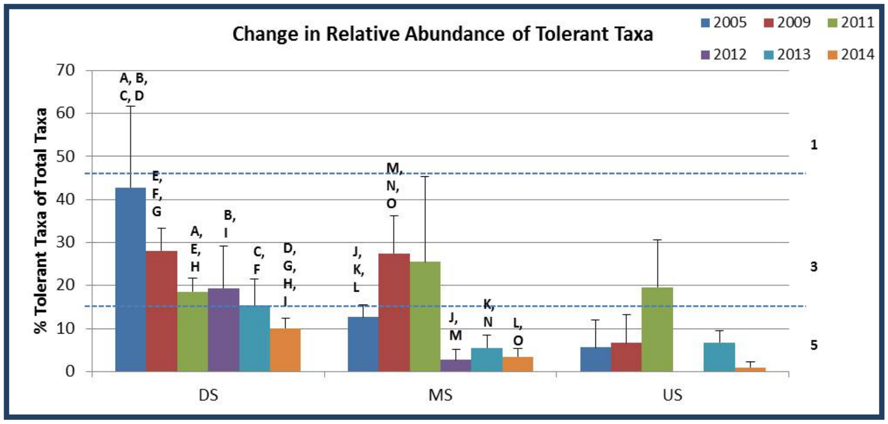
Figure 13.
Changes in dominance of the top (most abundant) taxon among Whychus Creek sampling sites. Ranges for IBI scoring (5, 3, 1) are indicated. Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Tolerance measures

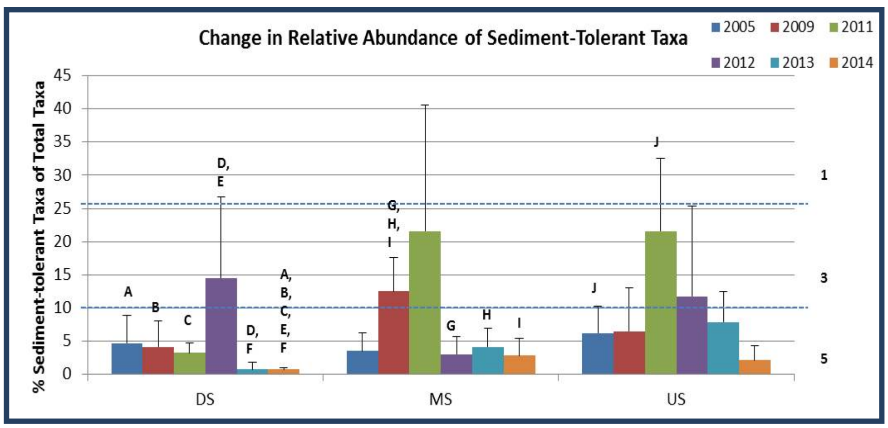
Metrics in the ORDEQ IBI that relate to tolerance are based on relative abundance and include % tolerant taxa (i.e. taxa tolerant to high levels of disturbance and pollution) and % sediment-tolerant taxa as well as the weighted mean MHBI (Modified Hilsenhoff Biotic Index) value for the sample. MHBI is a metric that reflects a taxon's tolerance for organic pollution (Hilsenhoff, 1987); values range from 0-10, with lower values indicating increasing sensitivity. Tolerant taxa occur in both healthy and impaired habitats, but their ability to persist and even thrive under conditions with low dissolved oxygen, high turbidity, or heavy siltation means that as stressors increase and sensitive taxa drop out of the community, their relative abundance in the community increases.

The relative abundance of both tolerant taxa and sediment-tolerant taxa has decreased overall among all sampling reaches (Figure 14a, b), suggesting a response to improved stream conditions. With the exception of the relative abundance of tolerant taxa in downstream reaches, a pattern of increased relative abundance followed by a decrease within the past ten sampling years is seen for both metrics. In contrast, mean assemblage MHBI has increased significantly among downstream and mid-stream sites and decreased among upstream sites, suggesting that organic inputs may be a stressor in the lower reaches of the creek (Figure 14c), although thus far the mean weighted average of macroinvertebrate community MHBI has not been above the value that receives the lowest scaled score in the ORDEQ IBI (MHBI >5).

a. Changes in relative abundance of tolerant taxa



b. Changes in relative abundance of sediment-tolerant taxa



c. Changes in weighted mean of assemblage MHBI

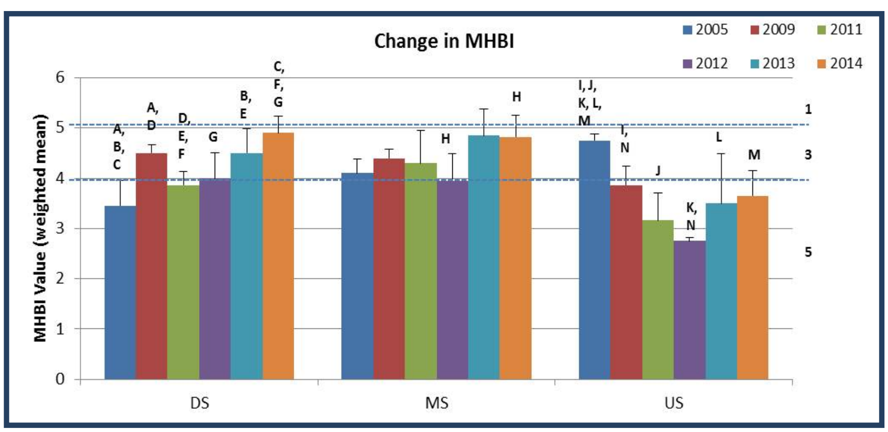


Figure 14.
Changes in tolerant taxa measures among Whychus Creek sampling sites. Ranges for IBI scoring (5, 3, 1) are indicated (note that these are negative metrics, i.e. a higher raw value receives a lower scaled IBI score). Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Functional Feeding Groups

Food sources and availability play a large role in structuring aquatic macroinvertebrate communities, so examination of different trophic guilds (i.e. functional feeding groups [FFGs]) can sometimes be informative. The proportion of different FFGs varies naturally based on stream order, reach location, and differences in allochthonous (terrestrial-derived, i.e. leaves, seeds, wood, carcasses, feces) and autochthonous (stream-derived, i.e. photosynthesis by primary producers, decomposition of dead organisms, feces) input of energy and nutrients into the stream (Vannote et al., 1980), but it can be further influenced by surrounding land uses, water diversions, and inputs of excess nutrients, sediment, or contaminants.

The main general functional feeding groups of aquatic macroinvertebrates include shredders, collectors, scrapers, and predators. Shredders (SH) rely on terrestrial organic input such as leaf litter, and are often a dominant FFG in headwater reaches, especially where riparian shading is significant. Collectors include both filterers (CF) and gatherers (CG), which take in fine particulate matter from the water column or sediment. Collectors are generally co-dominant with SH in headwater reaches, and then increase in dominance with increasing stream size. Scrapers (SC), also called grazers, rasp algae off of surfaces, and are thought to become dominant where primary production is maximized. The proportion of predators (PR) as a component of the stream community remains relatively steady with stream order.

Metrics based on FFGs have had differing degrees of utility in aquatic macroinvertebrate studies (Plafkin et al., 1989; Barbour et al., 1996; Fore et al., 1996), but as we had not previously investigated these traits in the Whychus Creek macroinvertebrate assemblages it seemed worthwhile to explore their utility. Overall, the greatest proportion of total taxa in each sampling year was comprised of predators, followed in decreasing order of relative richness by collector-gatherers, scrapers, shredders, and collector-filterers (Figure 15). A high percentage of scrapers to filterers generally reflects an unbalanced community and suggests a higher proportion of CPOM (coarse particulate organic material) compared to FPOM (fine particulate organic material). However, this higher scraper relative richness, combined with lower shredder relative richness, may reflect the stream's setting; as a high-elevation stream in an arid region with limited shading vegetation, greater sunlight penetration could favor primary producers and thus provide more algae for scrapers, with fewer allochthonous inputs for shredders.

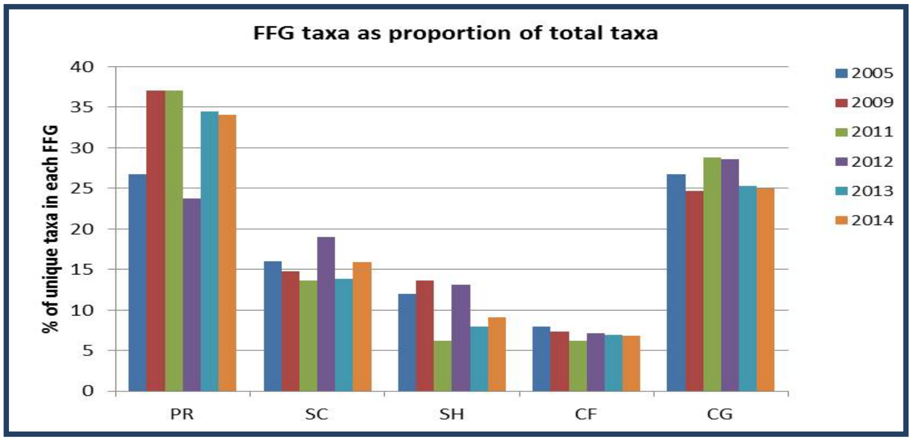


Figure 15.
Proportion of total taxa collected across all Whychus Creek sampling reaches comprised by each FFG. PR, predator; SC, scraper; SH, shredder; CF, collector-filterer; CG, collector-gatherer.

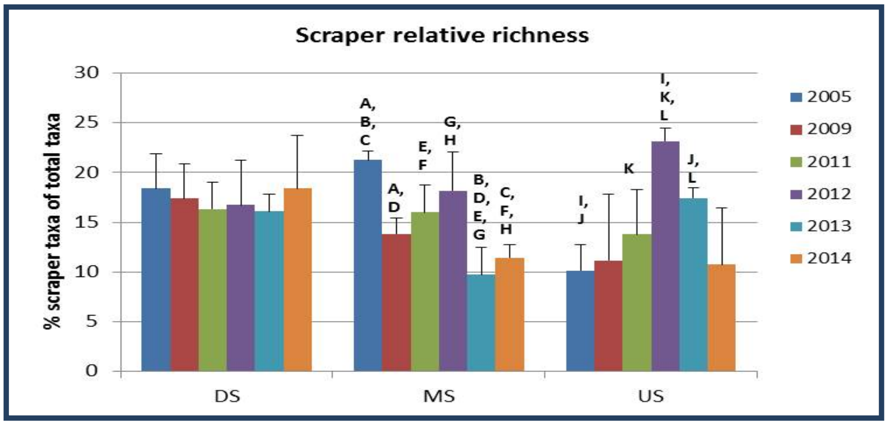
The proportion of collectors in the total macroinvertebrate community remained relatively constant for both filterers and gatherers, as did the proportion of scrapers. The proportion of predators appeared to increase overall, while shredder relative richness varied more from year to year, with a trend towards an overall decrease. Collectors are generalist feeders with a fairly broad food range and thus can be more tolerant of disturbances that might alter food availability; shredders are more specialized and less tolerant of disturbance, so their greater annual variation may be a response to changing stream conditions. The increase in predator relative diversity may also reflect an improved prey base. We examined changes in FFG proportions more closely by assessing changes in relative richness and relative abundance of each FFG among different sampling reaches (see below).

Scrapers

Scrapers, which include flatheaded mayflies, snails, and some caddisflies, are herbivores and detritivores that feed on periphyton and associated microflora and microfauna on organic or mineral substrates. In Whychus Creek, although the overall relative richness of SC taxa was similar across all sampling reaches, and the proportion of SC taxa in the downstream reaches remained about the same across time (Figure 16a), the relative abundance of scrapers in downstream reaches decreased substantially (Figure 16b). Both the relative richness and relative abundance of scrapers decreased in the mid-stream reaches, while the opposite was seen in the upstream reaches, although the increase in upstream SC relative richness that occurred through 2012 has decreased again significantly.

Scraper response to stream disturbance is variable (Resh, 1994; Hannaford and Resh, 1995; Fore et al., 1996). A high proportion of scrapers indicates a good supply of periphyton, although filamentous algae (which can increase with organic pollution) and mosses can interfere with their feeding activity. Scrapers rely on algae, which in turn require enough light penetration into the stream to grow, and so they are generally less dominant in deeper higher-order streams where there is less light penetration, but increase with increasing stream width and light penetration (Vannote et al., 1980). Since Whychus Creek lacks abundant shading vegetation, the increase in abundance and diversity in the upstream reaches may reflect greater light penetration and periphyton growth due to decreased sediment, as sustained changes in the macroinvertebrate assemblage have included an increase in taxa with lower fine sediment tolerances.

a. Changes in relative richness of scraper taxa



b. Changes in relative abundance of scraper taxa

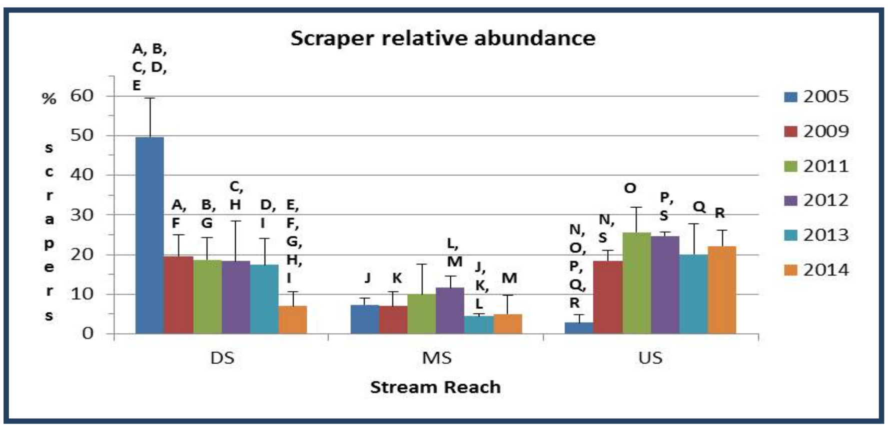
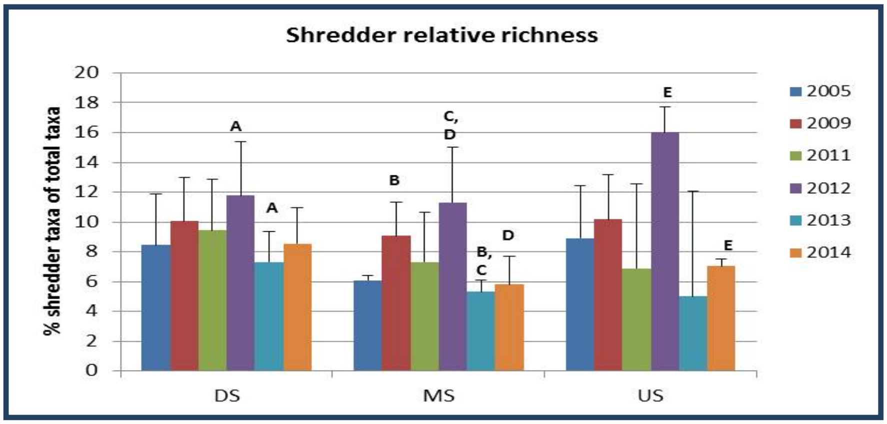


Figure 16.
Changes in scraper functional feeding group over time among Whychus Creek sampling sites. Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Shredders

Shredders, which include many stoneflies and some riffle beetles and caddisflies, feed on coarse living or dead organic matter. Because they process much of the allochthonous input into streams, shredders can be affected by surrounding land uses, especially in the riparian zone, and the proportion of shredders in a sample is expected to decrease with increasing disturbance. This can be due to both decreased input of material into the stream, as well as to the presence of toxins in the terrestrial area that can remain bound to leaf surfaces in the water. Relative abundance of shredders was low overall among stream reaches, and the relative richness varied greatly within and between years (Figure 18). However, a sustained decrease in SH relative abundance is evident among mid-reach sites, as well as a smaller but significant increase in SH relative abundance among downstream sites, and these may reflect changes in terrestrial inputs.

a. Changes in relative richness of shredder taxa



b. Changes in relative abundance of shredder taxa

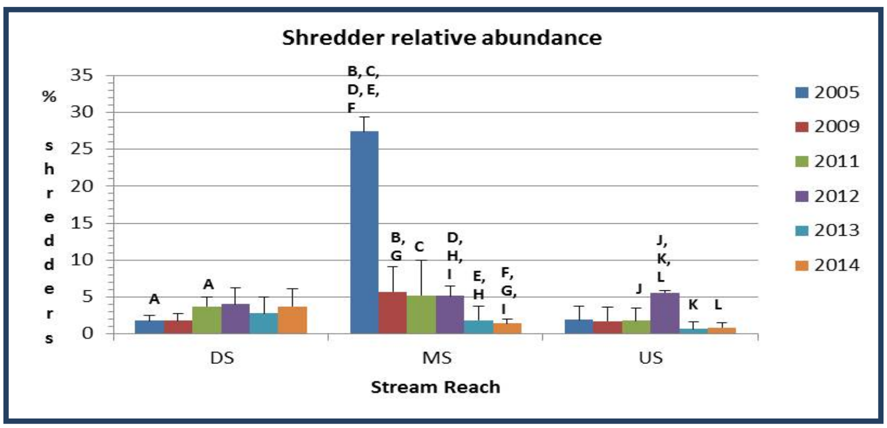
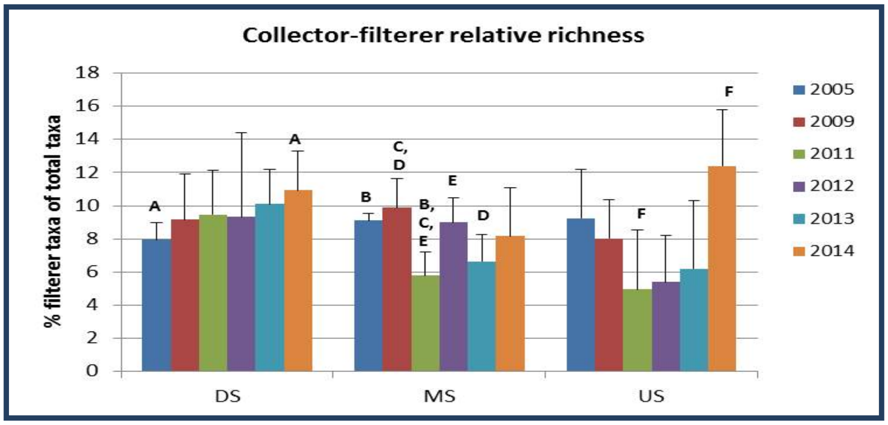


Figure 17.Changes in shredder functional feeding group over time among Whychus Creek sampling sites. Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Collector-filterers

Collector-filterers, which include black flies, net-spinning caddisflies, and fingernail clams, feed on suspended fine particles, and their response to disturbance varies. They have a broad food range and can thus be more tolerant of impacts that may affect food sources (Barbour et al., 1996), and because they can use filamentous algae as an attachment site, a high proportion of CF organisms may indicate organic enrichment. However, because some toxins can bind preferentially to small organic particles in aquatic systems, a decrease in CF may indicate the presence of contaminants. Excessive siltation can also interfere with their feeding abilities. Both the relative richness and relative abundance of CF varied greatly among all stream reaches (Figure 18), so any trends may not be highly informative. The greatest variation was seen among mid-stream reaches, which may reflect increased disturbance due to restoration projects. As both relative richness and abundance of CF in downstream reaches increased significantly between 2005 and 2015, this may reflect a real increase in FPOM material at these sites.

a. Changes in relative richness of collector-filterer taxa



b. Changes in relative abundance of collector-filterer taxa

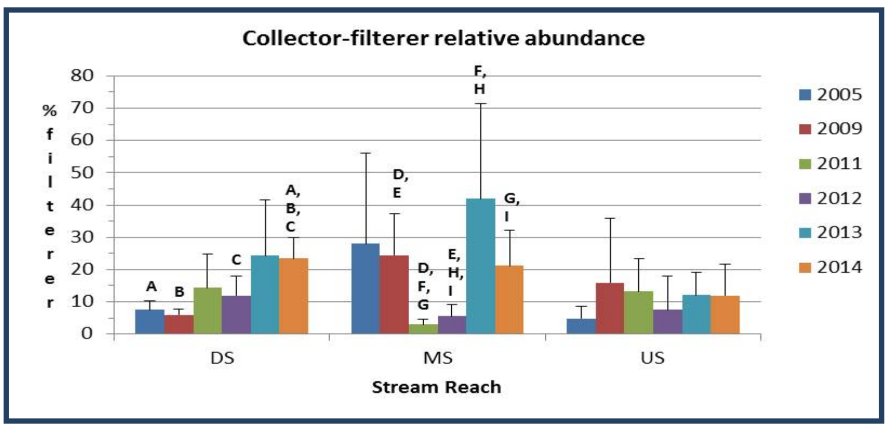
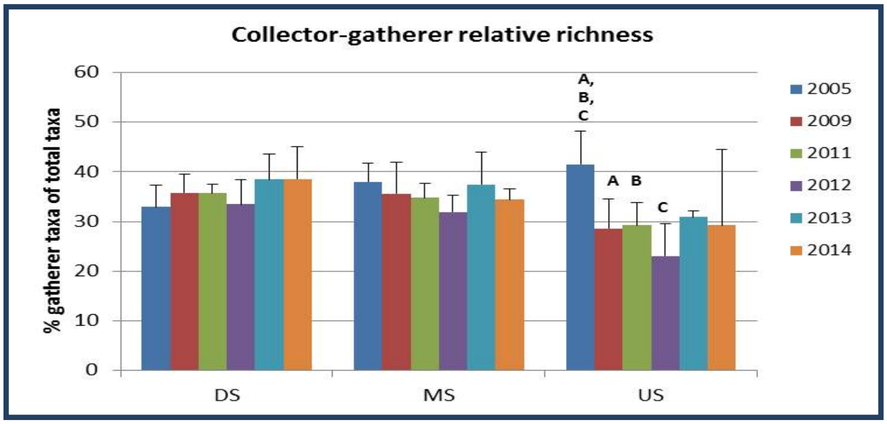


Figure 18.
Changes in collector-filterer functional feeding group over time among Whychus Creek sampling sites. Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Collector-gatherers

Collector-gatherers, which include many non-biting midge and riffle beetle larvae and caddisfly and mayfly nymphs, are detritivores and herbivores that feed in and on surface deposits. The response of this group to disturbance can also vary; organic enrichment may cause an increase in CG as less tolerant taxa are reduced, but toxins, especially those bound to organic particles, can reduce the CG population. Collectors are expected to be a dominant group in headwater and mid-sized streams (Vannote et al., 1980), and overall were the second most highly-represented FFG group among all Whychus Creek taxa. CG relative richness and relative abundance decreased significantly at upstream sites (Figure 19), which may be a reflection of the increase in abundance and diversity of sensitive taxa (i.e. EPT; see Figure 10) that also occurred at upstream sites. Little change was seen in CG relative richness between 2005 and 2014 among downstream and mid-stream reaches (Figure 19a), but the relative abundance of CG increased significantly across time within both sampling reaches (Figure 19b). However, initial relative abundances at these sites were lower than at upstream sites, and given that collectors in general are expected to be a dominant group in headwater, mid-size, and higher-order streams (Vannote et al. 1980) streams, the increase in CG relative abundance at lower reaches may reflect improved conditions.

Changes in relative richness of collector-gatherer taxa



Changes in relative abundance of collector-gatherer taxa

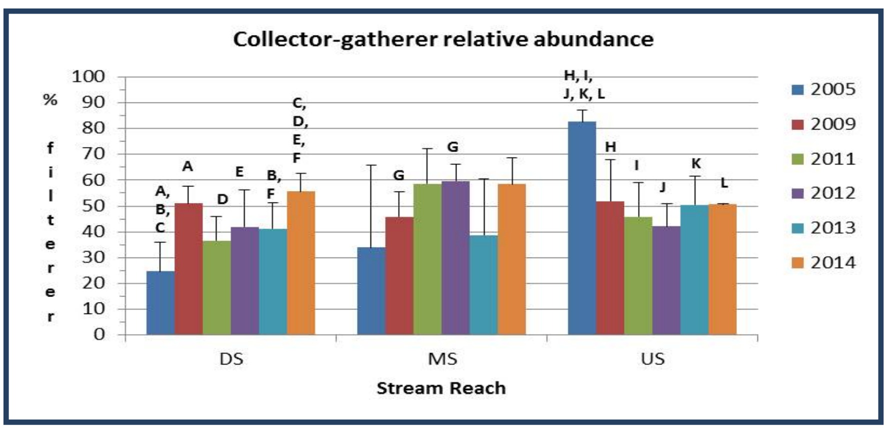
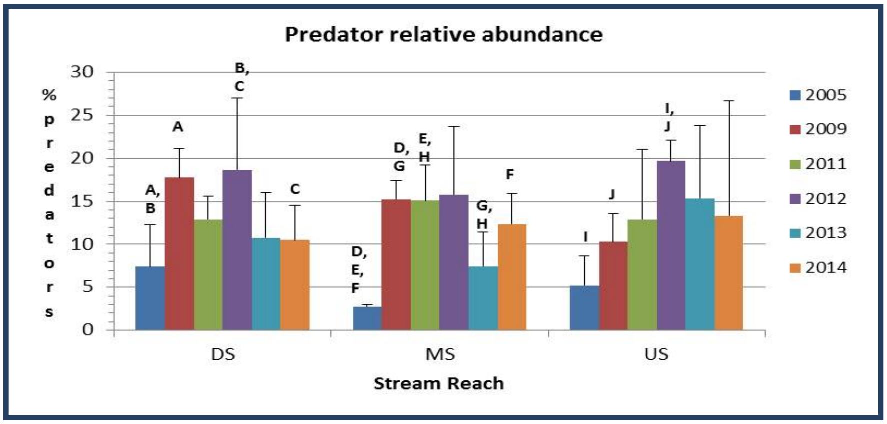


Figure 19.
Changes in collector-gatherer functional feeding group over time among Whychus Creek sampling sites. Downstream reach = RM 0.5 – 10.75; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Predators

Predators, which include all dragonflies and damselflies, and many stoneflies. caddisflies, and aquatic bugs and beetles, kill living organisms. Because they rely on an abundant food base, and because different life stages of a single taxon may require prey of different sizes, their relative abundance within a sample is expected to decrease with increasing disturbance. The relative abundance of predators varied greatly within and between sites among all reaches and years (Figure 20a), such that it is difficult to draw any conclusions, although there seems to be an overall upward trend. PR relative richness at mid-stream and upstream sites increased and then stabilized, while remaining relatively stable at downstream sites (Figure 20b). These results could indicate a sustained and possibly improving prey base.

a. Changes in relative abundance of predator taxa



b. Changes in relative richness of predator taxa

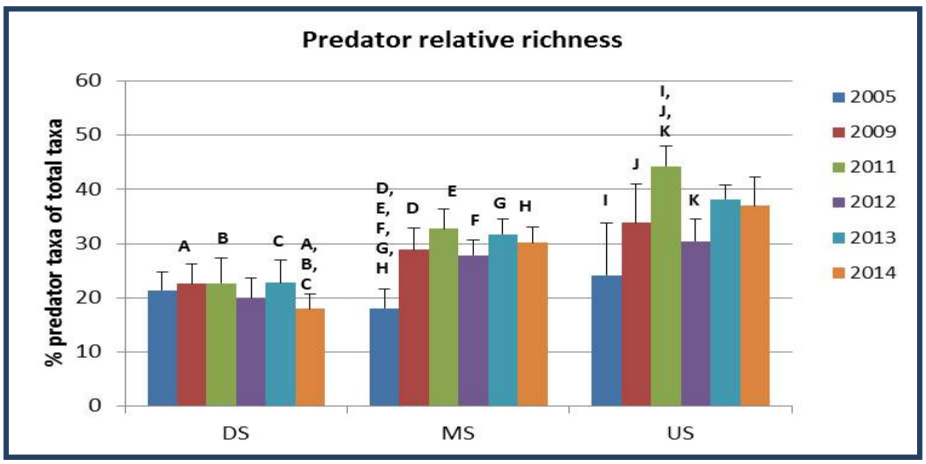


Figure 20.
Changes in predator functional feeding group over time among Whychus Creek sampling sites. Downstream reach = RM 0.5 – 9.5; mid-reach = RM 18.0 – 19.5; upstream reach = RM 23.5 – 30.25.

Community Similarity Analysis

Community composition

The number of unique taxa collected within each year has been similar, ranging from 76 to 83 (mean # taxa = 81 taxa $\pm$ 2.6). The lowest number of taxa (76) was found among all the sampling sites in 2005, but the slight increase in subsequent years is probably because several genera that were broken out into species in subsequent years were left at the genus level in 2005, especially *Epeorus* and *Rhyacophila*.

Since 2005, a total of 138 unique taxa have been collected among all sampling sites (see Appendix C for a complete taxa list). Thirty-three of these taxa were collected in every sampling year (for the mayfly genera *Ephemerella* and *Epeorus*, and the caddisfly genus *Rhyacophila*, members of the genus were present in each year but no single species in the genus occurred in all years). Five of the 81 taxa collected during the 2014 sampling season had never been collected in previous years: Gerridae (one at WC01900), Corixidae (one at each of two different upstream sites), *Caudatella edmundsi* (13 total among four upstream sites), Deuterophlebiidae (one at WC00150), and *Atrichopogon* (one at WC00600). The presence of gerrids (water striders) and corixids (water boatmen) in riffle samples at upstream sites is surprising, as these are tolerant families of aquatic bugs associated with slower-moving waters, but as only a single individual was collected at three different sites for both families combined, these were likely individuals taken by chance, probably from the quieter edge of the stream in a net set near the bank. Two of the five groups new in 2015 are sensitive taxa (*Caudatella edmundsi* is a sensitive species of spiny crawler mayfly, and this genus is a DEQ cool temperature and low sediment indicator; Deuterophlebiidae is a sensitive family of mountain midges), while *Atrichopogon* is a moderately tolerant genus of biting midge.

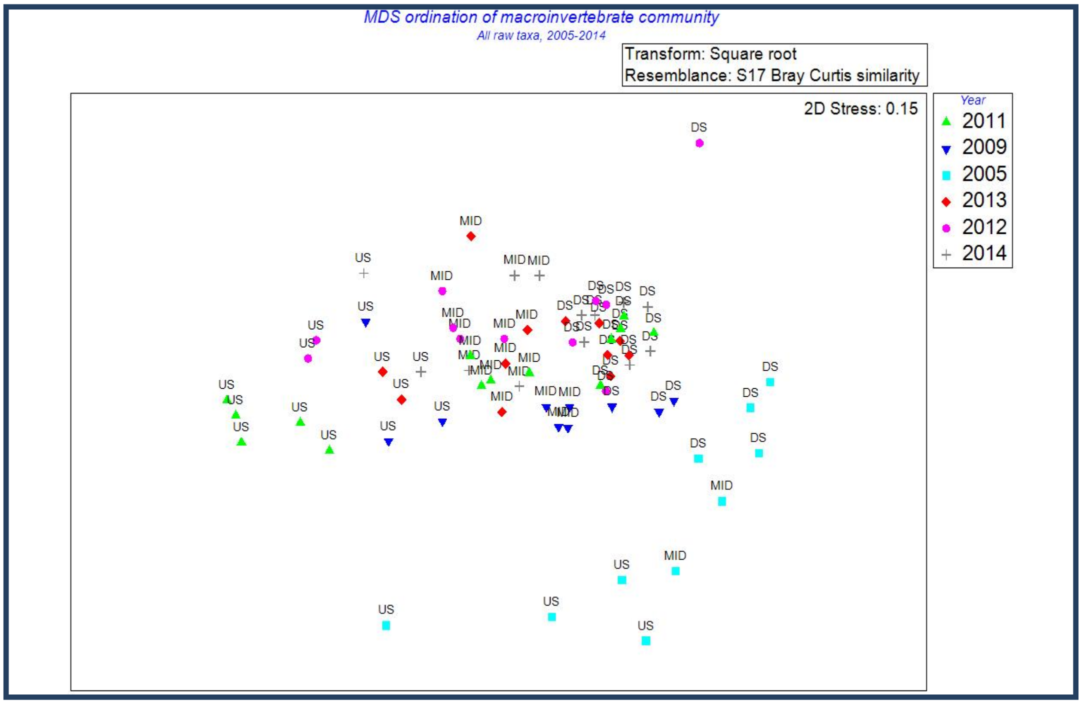
The need to subsample each composited site sample to 500 individuals during sorting and identification creates an inherent possibility of missing unique taxa, especially those occurring at very low abundances. However, the fact that anywhere from to 16-38% of all the taxa found within a given year have occurred at only one site indicates that rare taxa are found on a regular basis, and this continuing low level of addition to the taxa list with each year may be more suggestive of continuing changes within the macroinvertebrate community.

<u>MDS and SIMPER analysis</u>

In past years, analysis of a Bray-Curtis similarity matrix of square-root transformed abundance data has suggested a greater overall change in community composition from 2005 and 2009 compared to later years. However, several taxa identified only to genus in 2005 were identified to the level of multiple different species level in later years: *Ephemerella* and *Caudatella* mayflies*; Rhyacophila* and Leptophlebiidae caddisflies; and *Zapada* stoneflies. In order to be sure that changes in community composition observed after 2005 were not due solely to differing levels of taxonomic resolution, we did an MDS ordination of the data across all years using both the raw taxa (i.e. as identified in each sampling year), and the dataset with taxa for the above groups collapsed to genus (or in the case of Leptophlebiidae, family level). Duplicate samples taken for quality control in all years were omitted.

The results of both ordinations were similar (Figure 21), with all 2005 samples separated from all samples taken in subsequent years. As was also seen previously, sample clustering was influenced strongly by site location, with samples taken from similar reaches of the creek (upstream, mid-reach, or downstream) exhibiting the greatest similarity within and between years.

a. Macroinvertebrate assemblages at level of taxonomic resolution used in each year



b. Macroinvertebrate assemblages with relevant groups collapsed to level of taxonomic resolution used in 2005

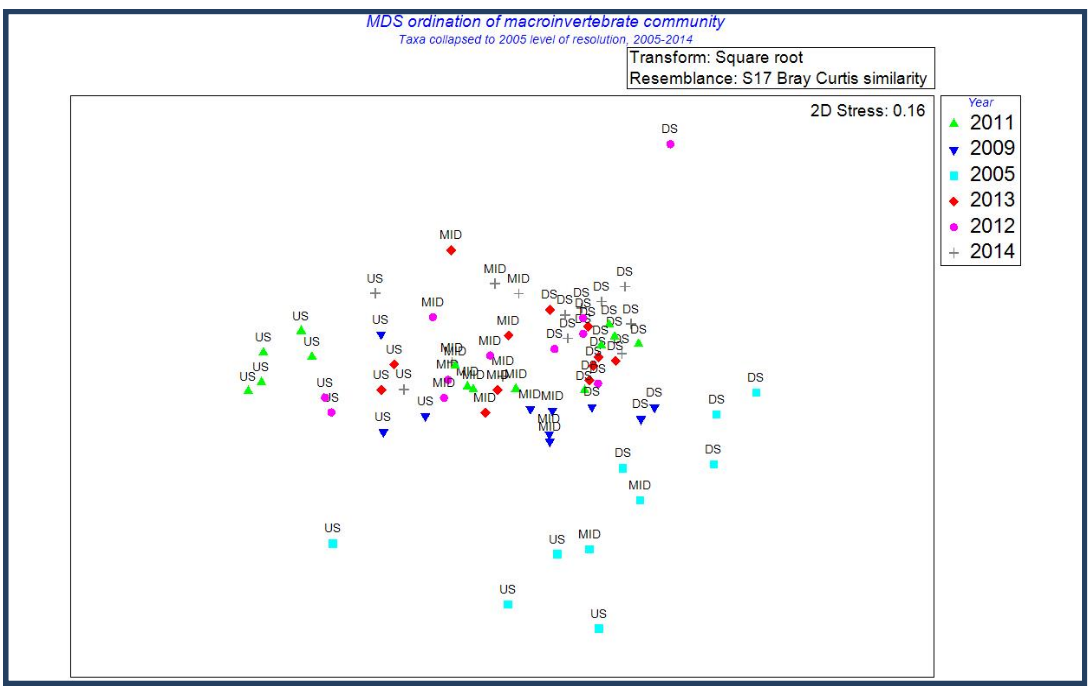


Figure 21.
MDS ordination of macroinvertebrate assemblages comparing the raw taxa (A) to taxa collapsed to the level of taxonomic resolution used in 2005 (B).Downstream reach (DS) = RM 0.5 – 9.5; mid-reach (MS) = RM 18.0 – 19.5; upstream reach(US) = RM 23.5 – 30.25.

Downstream sites

MDS ordination of all downstream reach sampling sites again showed a distinct separation between the 2005 downstream community compared to later years (Figure 22). When comparing the macroinvertebrate community present among all downstream sampling sites in different years, the lowest level of dissimilarity was seen between the community in 2013 and 2014 (35.6% dissimilar), and the greatest dissimilarities were seen between the community at downstream sites in 2005 and all other subsequent sampling years (Table 8). The downstream assemblage in 2005 was most similar to the community seen at the same sampling reaches in 2009 (47% dissimilar), and higher proportions of dissimilarity were seen in subsequent years.

The taxa that contributed the most to the dissimilarity with the 2005 community was similar between different years, and included *Zaitzevia* (tolerant riffle beetle genus; greater abundances in 2005), *Baetis* (common genus of small minnow mayfly; lower abundance in 2005), Chironominae (diverse subfamily of non-biting midge; lower abundance in 2005), *Brachycentrus* (sensitive genus of humpless case-maker caddisfly; lower abundance in 2005), and *Simulium* (moderately tolerant genus of blackfly; lower abundance in 2005).

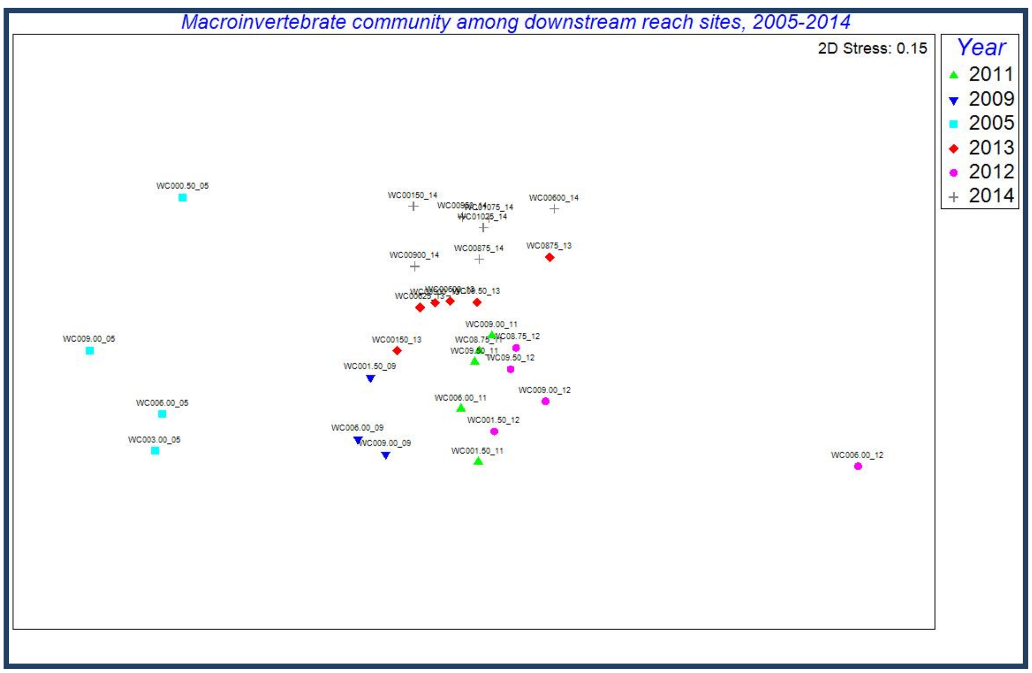


Figure 22.
MDS ordination of macroinvertebrate assemblages at all downstream reach sampling sites (RM 0.5 – 10.75).

Table 8.
Average percent dissimilarity in the macroinvertebrate community among all downstream reach sampling sites between years.

Year	2005	2009	2011	2012	2013	2014
2005	--					
2009	47.7%	---				
2011	51.9%	38.7%	---			
2012	56.4%	44.7%	37.7%	---		
2013	51.1%	38.8%	36.4%	41.5%	---	
2014	52.7%	45.0%	39.9%	44.9%	35.6%	---

Mid-stream sites

MDS ordination of all mid-stream reach sampling sites showed greater separation between the 2005 community compared to later years (Figure 23). When comparing the macroinvertebrate community present among all mid-stream sampling sites in different years, the lowest level of dissimilarity was seen between the community in 2011 and 2012 (34.8%% dissimilar), and the greatest dissimilarities were again seen between the community at mid-stream sites in 2005 and all other subsequent sampling years (Table 9). The mid-stream assemblage in 2005 was most similar to the community seen at the same sampling reaches in 2009, but the dissimilarity was still large (50% dissimilar), and higher proportions of dissimilarity were seen in subsequent years.

The taxa that contributed the most to the dissimilarity with the 2005 community were similar between different years, and included total *Baetis* (common genus of small minnow mayfly; lower abundance in 2005), total *Zapada* (relatively sensitive genus of small brown stonefly; greater abundance in 2005); *Simulium* (moderately tolerant genus of blackfly; greater abundance in 2005); and Chironominae (diverse subfamily of non-biting midge; greater abundance in 2005).

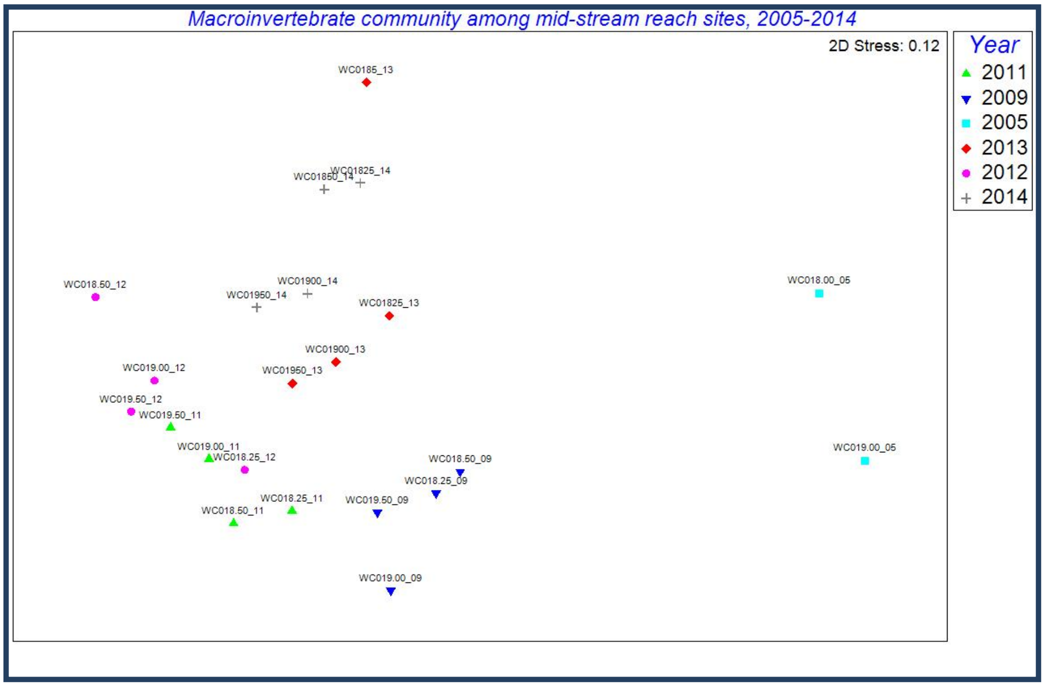


Figure 23.
MDS ordination of macroinvertebrate assemblages at all mid-stream reach sampling sites (RM 18.25 – 19.5).

Table 9.
Average dissimilarity in the macroinvertebrate community among all mid-stream reach sampling sites between years.

Year	2005	2009	2011	2012	2013	2014
2005	--					
2009	50.0%	---				
2011	60.3%	39.6%	---			
2012	59.5%	45.6%	34.8%	---		
2013	54.9%	45.7%	41.1%	43.2%	---	
2014	56.6%	45.1%	43.3%	41.5%	39.7%	---

Upstream sites

Differences among upstream sites may be exacerbated by the fact this reach contained the fewest sampling sites overall. However, MDS ordination of all upstream reach sampling sites showed a large separation between the 2005 community as well as the 2011 community compared to later years (Figure 24). When comparing the macroinvertebrate community present among all upstream sampling sites in different years, the lowest level of dissimilarity was seen between the community in 2009 and 2013 (39% dissimilar), and the greatest dissimilarities were again seen between the community at mid-stream sites in 2005 and all other subsequent sampling years (Table 10). The upstream assemblage in 2005 was most similar to the community seen at the same sampling reaches in 2009, but the percent dissimilarity was large (64.2% dissimilar).

Taxa that contributed most to the dissimilarity with the 2005 community were similar between different years, and included *Rhithrogena* (sensitive flatheaded mayfly genus; lower in 2005), *Suwallia* (sensitive genus of green stonefly; lower abundance in 2005), Orthocladiinae (somewhat sensitive subfamily of non-biting midges; greater abundance in 2005), *Prosimulium* and *Simulium* (black fly genera; lower abundance in 2005), and Chironominae (diverse subfamily of non-biting midge; greater abundance in 2005).

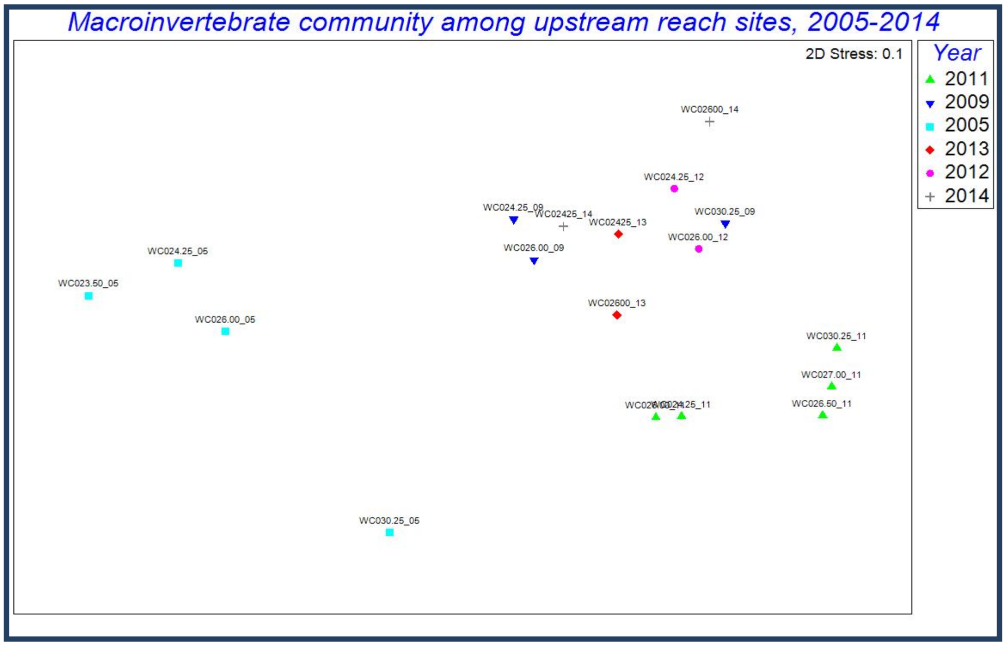


Figure 24.
MDS ordination of macroinvertebrate assemblages at all upstream reach sampling sites (RM 23.5 – 30.25).

Table 10.
Average dissimilarity in the macroinvertebrate community among all upstream reach sampling sites between years.

Year	2005	2009	2011	2012	2013	2014
2005	--					
2009	64.2%	---				
2011	74.6%	53.7%	---			
2012	73.5%	44.6%	49.8%	---		
2013	65.8%	39%	48.5%	40.5%	---	
2014	68%	42.3%	56.1%	44.9%	40.2%	---

Several of the taxa that discriminated between the macroinvertebrate assemblage in 2005 vs. later years are OR DEQ indicators. *Zaitzevia*, an indicator for both warm temperatures and high sediment, had greater abundances in 2005 at downstream sites; *Baetis*, a low sediment indicator, had lower abundances in 2005 among downstream and mid-stream sites; and *Rhithrogena*, a low sediment indicator, along with *Prosimulium*, a cool temperature indicator, had lower abundances among upstream sites in 2005. These results suggest a community changing in response to lower temperatures and decreased fine sediments resulting from restoration projects, similar to the findings for missing and replacement taxa and increaser and decreaser taxa.

Conclusions

Recovery trajectories of macroinvertebrate communities following removal of stressors from a stream system are often complex and may be non-linear (Clements et al., 2010), and their assessment is complicated by annual and seasonal variations and sporadic events such as flooding or drought, as well as the presence of continued stressors other than the one(s) addressed. The potential introduction of novel stressors in the system, both within the reach or stream and/or the larger catchment or watershed, can also influence recovery. Longer-term monitoring programs that have baseline, pre-project community data for comparison and continue for multiple years provide the best opportunity for seeing real differences in the biotic community following restoration.

Data collected from reaches along Whychus Creek since 2005 show definite changes within the macroinvertebrate community that may be attributed to the effects of restoration. Site PREDATOR and IBI scores have not shown a significant improvement in response to stream restoration activities, apart from a slight trend for increase in mean IBI scores among upstream sites, and this remains a confounding factor. Given the positive changes observed in several of the individual ORDEQ IBI metrics, and the better biological conditions indicated overall by the Grande Ronde IBI, the possibility remains that the parameters of established models are not ideal for this system, especially as PREDATOR scores were not found to be substantially different when the same taxa data for all sites was run through the PREDATOR model by DEQ staff in 2014. It is also important to remember that there are likely other stressors present in the watershed that have not been remediated by recent restoration activities.

However, examination of individual taxonomic and ecological traits of the community as well as the PREDATOR-generated increaser/decreaser taxa and missing/replacement taxa clearly shows that the macroinvertebrate assemblages at the sampling sites have changed over the years. Further, the direction of change reflects lower temperature and fine sediment conditions, indicating a measureable response to restoration activities. Results from multiple different analyses point towards a community changing from one with a greater number of tolerant and sediment-tolerant taxa to one comprised of more sensitive and sediment-intolerant taxa. Sediment and temperature optima were lower among increaser taxa and among replacement taxa compared to decreaser and missing taxa, with the exception of 2005, and mean sediment and temperature optima have decreased among the entire community collected at downstream, mid-stream, and upstream sites. Sensitive taxa measures such as EPT (mayfly, caddisfly, and stonefly taxa) have also shown improvement. While some metrics continue to have much more variation and less of a clear trend, and analysis of functional feeding groups was largely uninformative, MDS and SIMPER analysis of the macroinvertebrate assemblages show a community that has changed greatly since 2005, and many of the taxa that contribute the most to these dissimilarities between the 2005 samples and those in later years are DEQ indicator taxa that again reflect lower temperature and sediment conditions in more recent sampling years.

Thus, although standard assessments such as PREDATOR and IBI have not indicated substantial improvements in biolocgial conditions in Whychus Creek, analysis of multiple individual community attributes and characteristics strongly suggests a community whose composition has changed in response to changing stream conditions, that reflects a greater abundance and/or diversity of taxa that thrive in conditions with cooler temperatures and lower percent fine sediments.

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APPENDIX A Whychus Creek field data sheet

Site ID: Date: Sampled by:

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

Sample Information:

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

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

Total # sample jars _______

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

A = absent **B** = on bank **C** = ≤ 30 ft from bank **D** = > 30 ft from bank

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

Qualitative observations:

<u>Water odors</u>: none **/** organic **/** rotten eggs **/** fishy **/** chlorine **/** petroleum **/** other (describe):

<u>Water appearance</u>: clear **/** turbid **/** milky **/** dark brown **/** foamy **/** oily sheen **/** other (describe):

<u>Dominant land use</u>: Forest **/** agriculture (crops / pasture) **/** urban (industrial / residential) **/** other:

<u>Extent of algae covering submerged materials:</u> none **/** 1-25% **/** 25-50% **/** 50-75% **/** 75-100%

<u>Type of algae</u>: none **/** filamentous (i.e. strands >2") **/** close-growing **/** floating clumps

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

Substrate

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

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

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

Additional notes or observations (including other wildlife noted):

APPENDIX B Map of sampling sites throughout Whychus Creek

APPENDIX C Macroinvertebrate taxa list for Whychus Creek, 2005-2014

Phylum or subphylum	Class or Subclass	Order	Family	Genus	Species	2005	2009	2011	2012	2013	2014
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	Atrichopogon							√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Dasyhelea				√	√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Forcipomyia				√			
Arthropoda	Insecta	Diptera	Blephariceridae	Bibiocephala					√		

Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicera		√	√	√		√	√
Arthropoda	Insecta	Diptera	Deuterophlebiidae								√
Arthropoda	Insecta	Diptera	Simuliidae	Prosimulium		√	√	√	√		√
Arthropoda	Insecta	Diptera	Simuliidae	Simulium		√	√	√	√	√	√
Arthropoda	Insecta	Diptera	Ephydridae			√	√				
Arthropoda	Insecta	Diptera	Psychodidae	Pericoma		√	√			√	
Arthropoda	Insecta	Diptera	Psychodidae	Maruina			√			√	
Arthropoda	Insecta	Diptera	Tabanidae				√	√			
Arthropoda	Insecta	Diptera	Sciomyzidae							√	
Arthropoda	Insecta	Ephemeroptera	Baetidae	Acentrella		√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Baetidae	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	edmundsi						√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	hystrix	√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	spinifera	√		√	√	√	√

Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	flavilinea				√		
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	doddsi					√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	coloradensis		√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	albertae			√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	deceptivus/ hesperus			√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	grandis		√	√	√		√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Epeorus	longimanus		√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Rhithrogena		√	√	√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygmula		√	√	√	√	√	
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygma					√		
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	Tricorythodes		√					
Arthropoda	Insecta	Ephemeroptera	Leptophlebiidae	Paraleptophlebia		√	√	√	√	√	√
Arthropoda	Insecta	Hemiptera	Corixidae								√
Arthropoda	Insecta	Hemiptera	Gerridae								√
Arthropoda	Insecta	Megaloptera	Sialidae	Sialis		√					
Arthropoda	Insecta	Plecoptera	Perlidae			√	√	√			
Arthropoda	Insecta	Plecoptera	Perlidae	Calineuria	californica	√	√				
Arthropoda	Insecta	Plecoptera	Perlidae	Hesperoperla		√					
Arthropoda	Insecta	Plecoptera	Perlidae	Doroneuria						√	
Arthropoda	Insecta	Plecoptera	Perlodidae			√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Isoperla			√	√			√
Arthropoda	Insecta	Plecoptera	Perlodidae	Megarcys			√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Rickera	sorpta		√	√			√

Arthropoda	Insecta	Plecoptera	Perlodidae	Kogotus		√				√	√
Arthropoda	Insecta	Plecoptera	Perlodidae	Skwala		√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Paraperla		√			√		√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Suwallia			√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Chloroperlidae	Sweltsa		√	√	√		√	√
Arthropoda	Insecta	Plecoptera	Leuctridae				√	√			
Arthropoda	Insecta	Plecoptera	Leuctridae	Despaxia	augusta		√				
Arthropoda	Insecta	Plecoptera	Nemouridae	Amphinemura					√		
Arthropoda	Insecta	Plecoptera	Nemouridae	Visoka	cataractae	√	√	√	√		√
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada		√	√	√	√		
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	cinctipes		√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	columbiana		√		√	√	
Arthropoda	Insecta	Plecoptera	Nemouridae	Zapada	oregonensis			√			√
Arthropoda	Insecta	Plecoptera	Pteronarcyidae	Pteronarcys		√	√	√	√	√	√
Arthropoda	Insecta	Plecoptera	Peltoperlidae	Yoraperla		√					
Arthropoda	Insecta	Plecoptera	Capniidae			√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Apataniidae	Pedomoecus			√			√	
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Agapetus		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Glossosomatidae	Glossosoma		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Arctopsyche	grandis	√	√				
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsyche		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Parapsyche	elsis		√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Brachycentridae	Micrasema		√	√	√	√		√
Arthropoda	Insecta	Trichoptera	Brachycentridae	Brachycentrus	americanus	√	√	√	√	√	√

Arthropoda	Insecta	Trichoptera	Helicopsychidae	Helicopsyche		√					
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila		√	√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Angelita Gr.			√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	arnaudi		√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	atrata complex					√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Betteni Gr.		√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Brunnea/ Vemna Gr.		√	√	√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	coloradensis						√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Hyalinata Gr.		√	√			√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	narvae		√	√			
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Nevadensis Gr.			√			
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	grandis		√				
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Vagrita Gr.		√	√			√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	valuma		√	√	√		
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	vetina complex					√	
Arthropoda	Insecta	Trichoptera	Sericostomatidae	Gumaga					√		
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Agraylea		√					
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptila		√	√	√		√	√
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Metrichia		√					
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Ochrotrichia		√	√			√	
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Stactobiella				√	√		
Arthropoda	Insecta	Trichoptera	Lepidostomatidae	Lepidostoma			√			√	√

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

Native Fish Monitoring in Whychus Creek

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Abstract

Restoration actions on Whychus Creek aim to restore the stream habitat, flows, fish passage, and water quality necessary to support self-sustaining populations of reintroduced Chinook salmon (*Oncorhyncus tshawytscha*), anadromous steelhead and resident redband trout (*Oncorhyncus mykiss*), and bull trout. Hundreds of thousands of steelhead and Chinook fry and thousands of smolts have been released annually since 2007 and 2009, respectively. The Whychus Creek Monitoring Plan identified fish populations monitored annually by Portland General Electric as a biological indicator of restoration effectiveness. But, fish population data are confounded by the ongoing release of hundreds of thousands of juvenile steelhead and salmon annually in concert with implementation or recent completion of multiple restoration projects along the creek, limiting the applicability of these data to evaluate restoration effectiveness. While recognizing this limitation, the Upper Deschutes Watershed Council summarizes PGE's and other available fish monitoring data for Whychus Creek annually to track the status and trends of fish populations. Although *O. mykiss* have accounted for the majority of fish caught in Whychus in every year since 2007, densities remain low with no significant increasing or decreasing trend. A 2015 genetic analysis of 2005 and 2013 Whychus *O. mykiss* populations showed the 2013 population dominated (75%) by Round Butte Hatchery stock, suggesting fry reintroduced into Whychus are residualizing and displacing natural origin fish. No Chinook smolts were released and therefore none captured in 2014; densities in previous years were similarly low to juvenile steelhead densities. Redd densities averaged 5 redds/km in 2014, slightly up from the 2013 estimate of 4.1 redds/km. No smolt outmigration estimates exist for Whychus due to no trapping in 2014 and low captures in previous years, but captures in previous years showed smolt outmigration occurring between March and May. Preliminary adult migration data show steelhead migration into the Upper Deschutes occurring from September to May and Chinook from May through July. Two returning adult Chinook and two steelhead ascended the Deschutes River in 2014; none ascended Whychus Creek. Improved information about Chinook, steelhead, and redband life histories and population dynamics in Whychus Creek 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 subbasin 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 population size and size-frequency distributions, spawning activity, and timing of spawning. PGE additionally monitors juvenile migration from Upper Deschutes tributaries into Lake Billy Chinook to estimate smolt production and establish migration timing and rate, 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
2014	310,900	4,498	-	-

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

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

Given these obstacles, available fish population data is of limited use over the short term as a biological indicator of restoration effectiveness in Whychus Creek. We anticipate native fish monitoring data 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 in 2017 and 2022, five and ten years after returning fish are first passed upstream of the hydroelectric project. These data will provide further insight into population dynamics and interactions between the two life histories, and between Round Butte and natural-origin stock. In the interim, UDWC continues to track PGE's native fish monitoring on Whychus Creek and summarize their findings in an annual technical report. As restoration projects are completed and additional fish population data become available, UDWC will 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 2015 Fisheries Monitoring reports, including juvenile Chinook and *O. mykiss* rearing densities (Madden *et al* 2015; Bennett *et al* 2015); *O. mykiss* redd densities (Madden *et al* 2015); Chinook and steelhead smolt outmigration (Hill & Quesada 2015); and adult Chinook and steelhead returns and migration (Wymore *et al* 2015). We compare 2014 data to 2007-2013 results; 2006 native fish monitoring data were collected using different methods and are not comparable to 2007-2014 data, and are therefore not considered in this report. We also summarize findings from the 2015 USFWS genetic analysis of *O. mykiss* in Whychus Creek (Adams *et al* 2015).

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

Chinook salmon

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

Sockeye salmon

Sockeye salmon (*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 juvenile sockeye rearing. Kokanee salmon, the landlocked form of sockeye, now utilize Lake Billy Chinook for rearing. These kokanee may be descended from Suttle Lake sockeye that were trapped behind the dams. Fies *et al* (1996) reported an observation of 11 kokanee salmon adults (spawners) in Whychus Creek downstream from Alder Springs during a survey in 1991. This may indicate a potential for anadromous sockeye salmon to spawn in Whychus Creek and rear in Lake Billy Chinook if runs are reestablished above the dams.

O. mykiss

Redband trout and summer steelhead trout are both classified as *Oncorynchus mykiss* (Behnke 2002). Redband exhibit a resident life history behavior and spend their entire life within a stream system, although they may migrate within the system. Small numbers of redband trout in the upper Deschutes River system 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 are reintroduced. 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 sampling at this site was discontinued in 2012 following diversion of the stream from the straightened channel, where prior sampling had occurred, into the restored meadow channel. 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 re-named reaches in 2014, assigning the Wolftree Reach as Reach 3. We use the original reach numbers as well as reach names to retain individual site data. PGE sampled Reaches 1, 2, 4, and 5 in 2014.

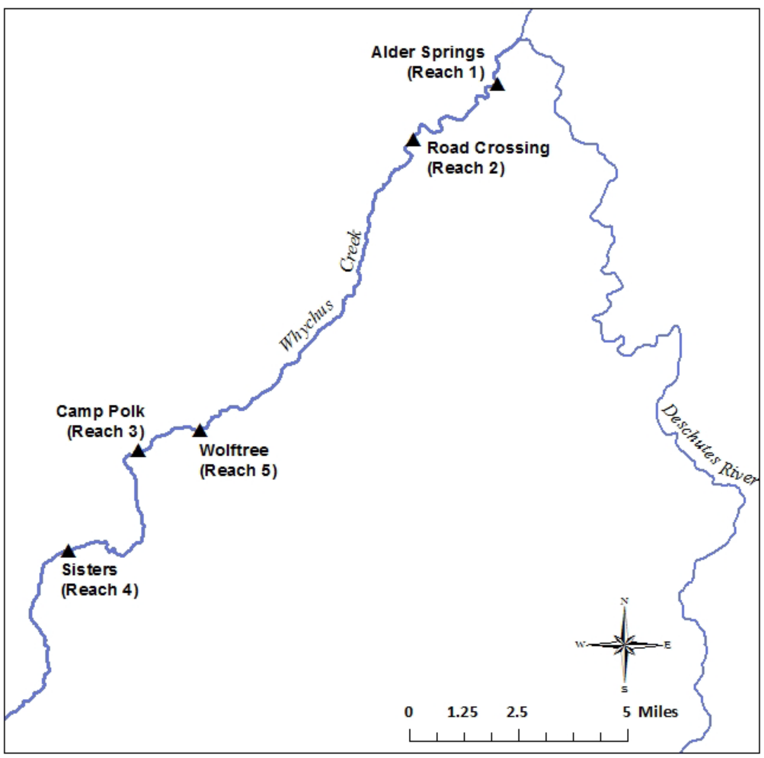

Figure 1.
Study reaches on Whychus Creek for fish population estimates. Alder Springs, Road 6360 Crossing and Sisters reaches (reaches 1, 2 and 4) have been surveyed annually since 2006; Camp Polk (Reach 3) was surveyed from 2006-2011; and Wolftree (Reach 5) was surveyed annually from 2009 through 2012 and in 2014. Reach 5 was not surveyed in 2013 due to high and turbid flows. Adapted from Madden *et al* 2015.

PGE conducted fish population sampling during the low flow period, from September 22-30, 2014. Study reach lengths ranged from 133-240 m, determined by the location of habitat characteristics allowing the secure placement of block nets. Block nets were situated above and below survey sections within each reach, with an additional net placed midsection to evaluate block net effectiveness and the mark-recapture sampling assumption of a closed population.

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

Mark-recapture electrofishing was conducted following protocols adapted from ODFW (Scheerer *et al* 2007), as described in Madden *et al* (2015). 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. No Chinook fry were released into Whychus in 2014 due to a shortage of Chinook fry from Round Butte Hatchery.

O. mykiss and spring Chinook smolt production

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

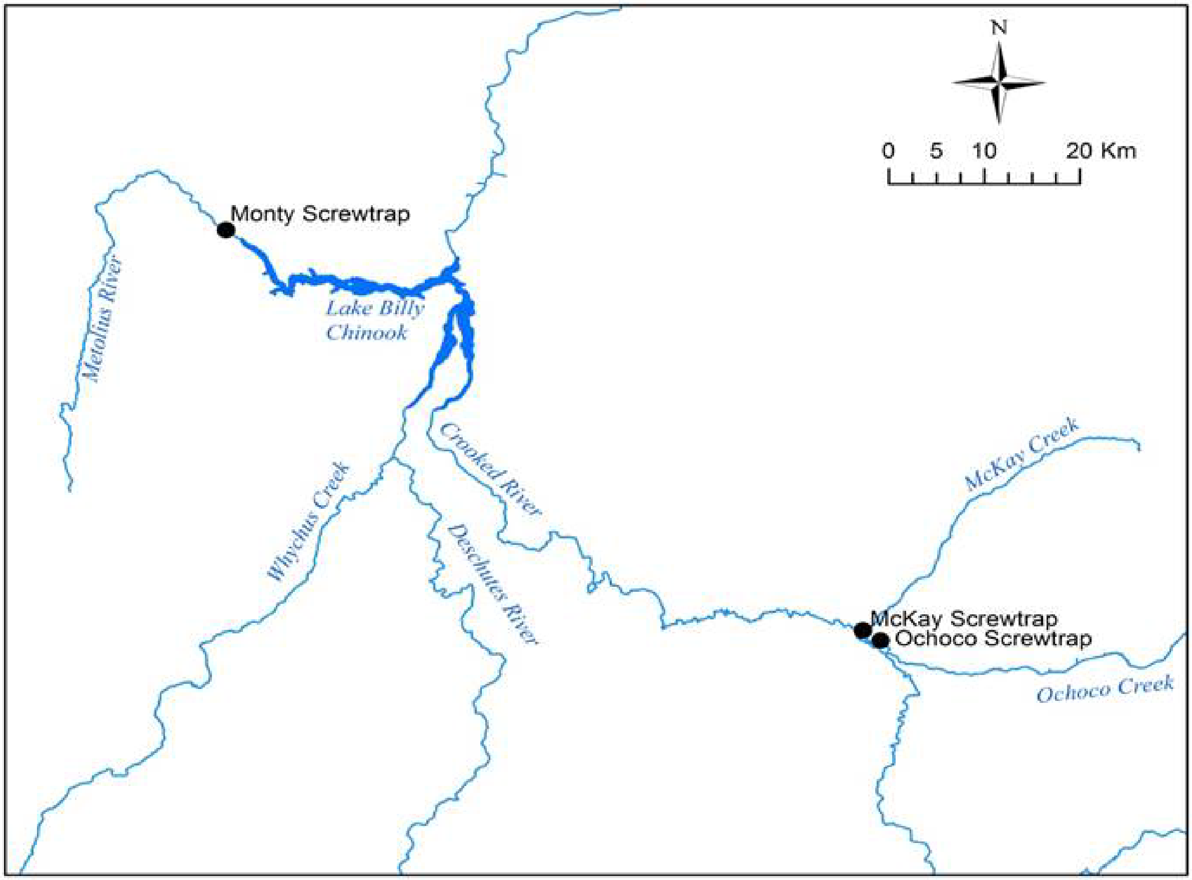

Figure 2.
2011-2013 trap locations. Reproduced with permission from Hill and Quesada 2014.

O. mykiss redd surveys

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

In 2010 PGE revised redd count methods to implement a spatially balanced rotating panel design recommended by American Fisheries Society Protocol. This design, similar to that used by the ODFW Coastal Salmonid Inventory Project (ODFW 2007), allows estimates of redds per kilometer and spawning distribution, reduces bias, and addresses shortcomings of the original index reach design, specifically, that index reaches may not be a reliable means to determine trends in spawning abundance because spawning site selection may not be consistent or predictable between years (Isaak and Thurow 2006). The rotating panel design incorporates two annually sampled index sites with two sites randomly selected from a predetermined set of reaches and sampled at regular, less frequent intervals (Gallagher et al. 2007). From 2010 to 2012 PGE sampled: 1) two 1-km index sites designated from the original ten reaches; 2) two sites randomly selected each year; and 3) four additional sites retained from the original ten, to help establish a population trend and to identify the temporal and spatial *O. mykiss* spawning distribution.

In 2013 and 2014 PGE and Deschutes National Forest conducted redband trout redd surveys at six sites on Whychus Creek (Figure 2): in the two index sites at Alder Springs (Reach 1, rkm 2) and immediately upstream of Camp Polk (Reach 6, rkm 27; Reach 8 prior to 2010); in one reach at rkm 21 (Reach 4) randomly selected in 2013; and at three additional sites, the restored meadow channel at Camp Polk (rkm 25; reach 5), Lewis Woodpecker Creek (reach 3), and Alder Springs Creek (Reach 2). In 2013 and 2014 the Camp Polk reach was surveyed in place of a second rotating panel design randomly selected site to allow continued 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.

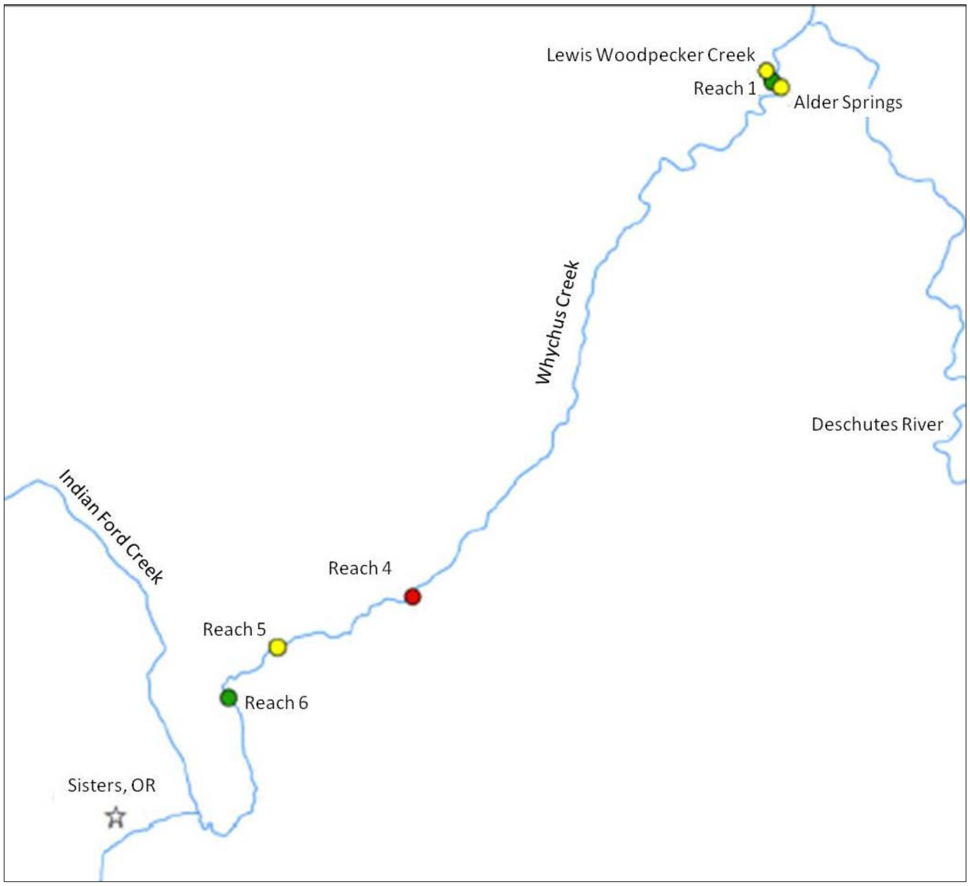

Figure 3.
Redband redds were counted in six reaches in 2014: two designated index reaches (green circles: reach 1, rkm 2; reach 6, rkm 27); one randomly selected reach (red circle: reach 4, rkm 21); and in three original reaches (yellow circles: reach 2, Alder Springs Creek; reach 3, Lewis Woodpecker Creek; and reach 5, Camp Polk). All reaches except reach 4 were surveyed from 2006-2014; reach 4 was surveyed in 2013 and 2014. 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 2014, all returning adult steelhead and Chinook salmon of known origin (identified by an intact adipose fin and a left or right maxillary clip) that returned to the Pelton Trap were tagged with two fluorescent green anchor tags, to distinguish them from conspecifics on spawning grounds, and passed upstream of Round Butte dam. Adults meeting criteria for condition, minimum size, maxillary clip and presence of a PIT tag were tagged with Juvenile Combined Acoustic Radio Telemetry (JCART) tags that emit both a radio and an acoustic signal to allow PGE biologists to track fish using either method.

Fish were tracked from fixed radio or acoustic telemetry stations at the upper end of the Deschutes, Crooked, and Metolius river arms of Lake Billy Chinook, as well as from 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 2014 the majority of fish captured in Whychus Creek were *O. mykiss* including both resident redband and released steelhead. One 88 mm bull trout was captured in the Alder Springs reach (Reach 1) during 2014 sampling, potentially indicating active bull trout spawning and rearing in this reach. Other species captured included brown trout, brook 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 eight years of sampling (Table 2, Figure 4). Estimated densities for any reach have rarely exceeded 40 *O. mykiss*/100m^2, compared to densities in McKay and Ochoco Creeks ranging from the 40s up to 281 *O. mykiss*/100m^2 at one site in 2014. Density estimates for 2014 remained relatively consistent with those for 2013. Size distribution of *O. mykiss* from 2007-2014, 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 2007-2014. Data from 2006 are not comparable due to differences in sampling methods, and thus are not included.

	O. mykiss/100m² and 95% confidence interval							
Reach	2007	2008	2009	2010	2011	2012	2013	2014
1 (Alder Springs)	48 ± 28	24 ± 24	12 ± 4	11 ± 4	24 ± 5	7 ± 2	27 ± 2	25 ± 4
2 (Road 6360)	25 ± 10	9 ± 3	24 ± 9	13 ± 4	15 ± 3	15 ± 3	31 ± 3	15 ± 6
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	35 ± 6
5 (Wolftree)	-	-	21 ±7	106 ± 29	42 ± 9	32 ± 9	-	31 ± 7
USFS site at TSID	-	-	2.4 1.5-4.0	-	-	-	-	

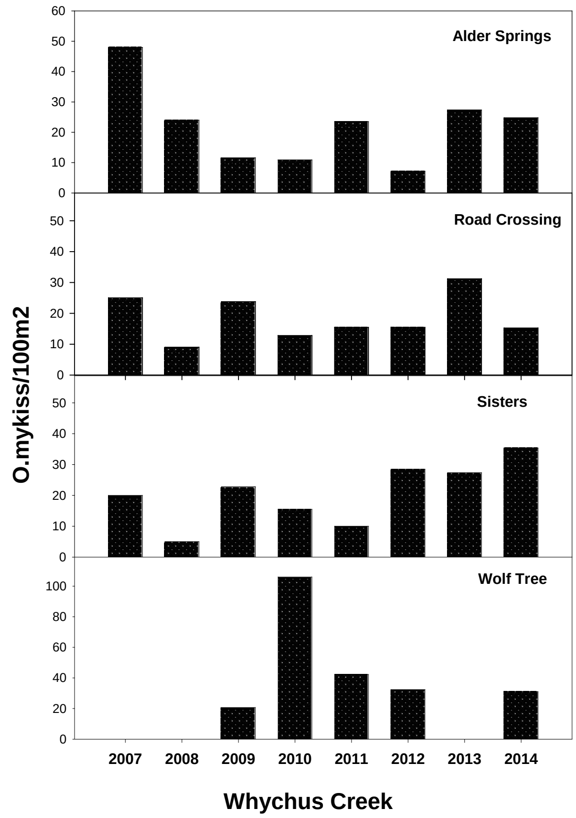

Figure 4.
O. mykiss densities at four Whychus Creek sites from 2007 to 2014. Surveys at Camp Polk were discontinued following diversion of Whychus Creek into the restored channel. Reproduced with permission from Madden *et al* 2015.

Spring Chinook juvenile density & size

Consistent with no Chinook fry having been released into Whychus Creek in 2014, no juvenile Chinook were captured during electrofishing surveys in Whychus Creek. Juvenile spring Chinook densities in Whychus Creek were low in 2013, but the total number caught was higher than in previous years (Table 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. No Chinook were released, and accordingly none captured, in Whychus Creek in 2014.

	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	--

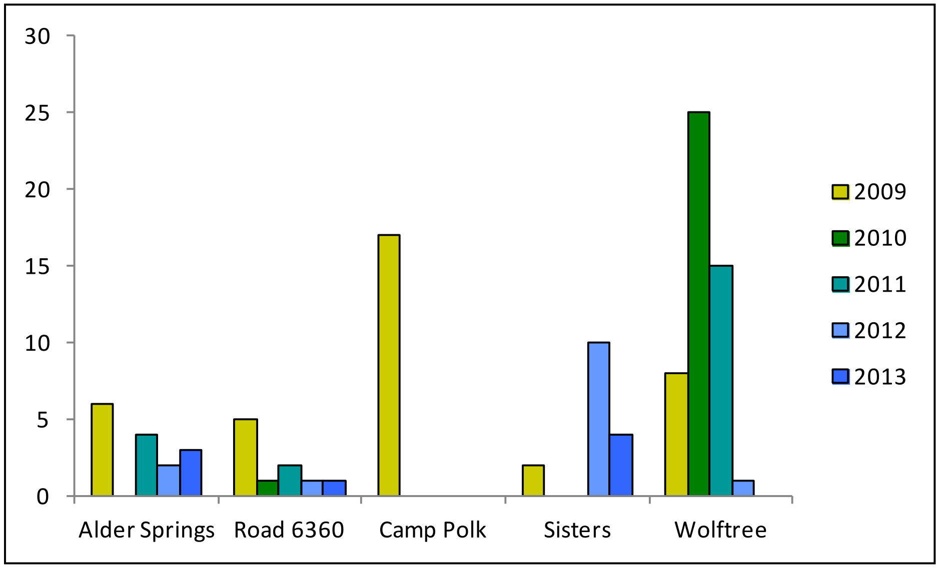

Figure 5.
Juvenile Spring Chinook densities in Whychus Creek in 2009-2013 estimated from mark-recapture surveys

Juvenile migration and smolt production

PGE did not deploy a trap for outmigrating smolts on Whychus Creek or the Deschutes River Arm of Lake Billy Chinook in 2014. In 2013 and 2012 fish captures at the Merwin trap deployed on the Deschutes River Arm were too low to generate a smolt production estimate for the Upper Deschutes and Whychus Creek. In 2013 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 2013, with captures from March through May.

Adult returns and migration

Fifty adult steelhead were captured at the Pelton Trap between September 2013 and April 2014, with captures peaking in October. Of these, 38 originated from fry releases or natural production (right maxillary-clipped) and 12 from smolt releases (left maxillary-clipped). This number is little more than a third of the 133 adult steelhead that returned to the Pelton Trap the previous year. Twenty-four adult Chinook salmon, two more than in 2013, returned to the Pelton Trap between early May and July, 2014, peaking in late May. Forty-nine of the adult steelhead and all 24 adult chinook were JCART-tagged and released upstream of Round Butte Dam. Two steelhead (4%) ascended the Deschutes River Arm from Lake Billy Chinook, of which one was detected in the middle Deschutes River near the confluence with Whychus, with a final location 0.4 km upstream of the confluence with Whychus Creek in the middle Deschutes River. One Chinook that had previously ascended the Crooked River Arm to Opal Springs Dam subsequently fell back below the dam and ascended the middle Deschutes River where it was last detected; one additional Chinook entered the Deschutes River Arm and ascended to the middle Deschutes River upstream of the confluence with Whychus Creek. No returning steelhead or Chinook ascended Whychus Creek in 2014. In 2013 one steelhead and three Chinook ascended Whychus Creek as high as Rimrock Ranch.

O. mykiss redd surveys

Surveyors identified a total of 42 redds in Whychus Creek in 2014 (Table 4; Figure 6). As in previous years, the majority of redds (81%) were located in reaches 1-3, in the vicinity of Alder Springs. Redds averaged 5/km in 2014, slightly higher than the 4.1 redd/km average from 2013, but lower than in previous years. Spawning in Whychus was recorded from April through July and peaked in June.

Table 4.
O. mykiss redd survey sites, reaches, and count data, 2006-2014.

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

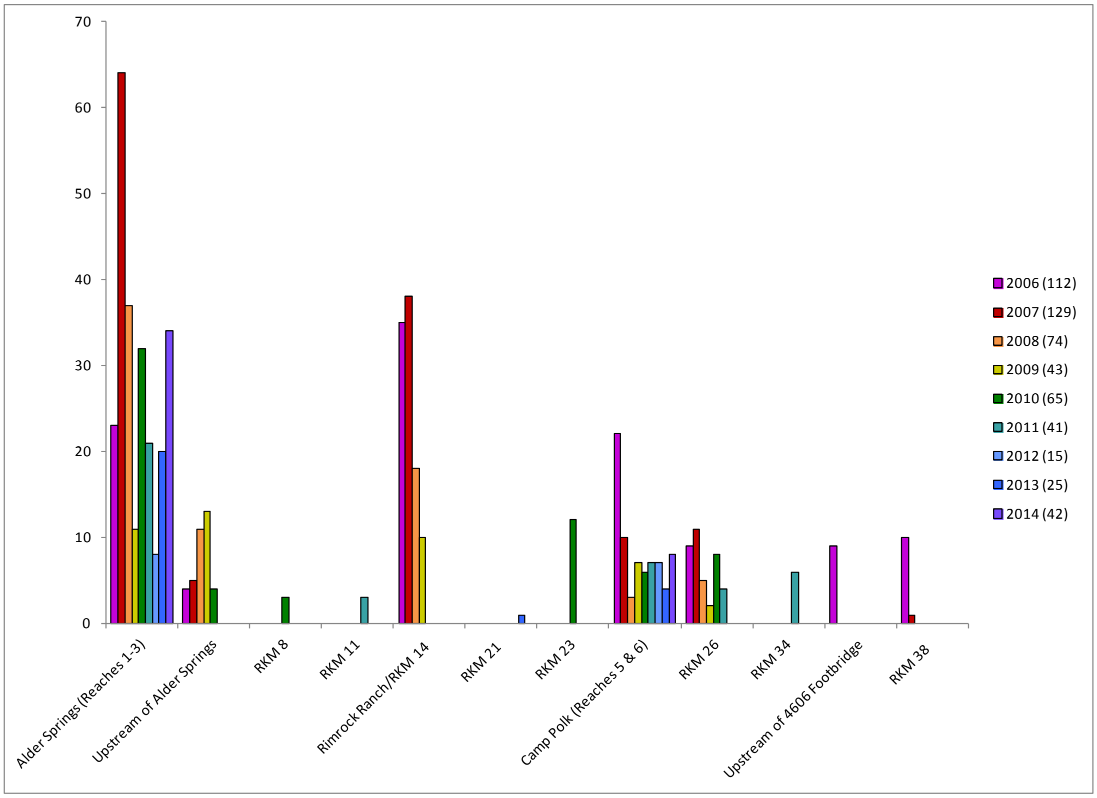

Figure 6.
Redband redds detected by site and year in Whychus Creek. Totals for each year are shown in the legend. Reach numbers correspond to 2013-2014 reaches.

Discussion

O. mykiss and Chinook Salmon population estimates

Despite annual releases averaging 276,750 fry and 4,270 smolts, far higher than in McKay and Ochoco Creeks, estimated juvenile *O. mykiss* density remains far lower in Whychus (half or less) than in the two Crooked River tributaries (Madden *et al* 2015). Densities in three of four reaches sampled in Whychus were approximately equivalent in 2014 and more or less consistent with 2013 numbers. On average, the highest densities have been observed at Alder Springs, followed by Sisters, the 6360 Road Crossing, and Wolftree. The relatively higher densities at Sisters suggest this reach may provide suitable rearing habitat; particularly so in combination with cooler summer stream temperatures in this reach than in downstream reaches. Although no density estimate was available for juvenile Chinook in Whychus Creek in 2014, Chinook densities in previous years were lower in Whychus Creek than in all other rivers and tributaries sampled (Metolius, Lake Creek, Ochoco Creek). Despite continued low *O. mykiss* densities in Whychus, 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.

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

Genetic analysis of 2005 (pre-reintroduction) and 2013 (post-reintroduction) Whychus Creek populations showed *O. mykiss* in Whychus in 2013 to be predominately (75%) descended from Round Butte Hatchery stock, with natural-origin *O. mykiss* largely absent (15%) from the 2013 Whychus Creek population. This finding indicates that Round Butte Hatchery *O. mykiss*, most likely fry released for reintroduction that residualized in the creek, have replaced natural-origin *O. mykiss* in Whychus Creek over the past two generations (Adams *et al* 2015), with implications for the contribution of a Whychus Creek population to Mid-Columbia steelhead recovery and for expression of genetic and life-history diversity in the Whychus Creek *O. mykiss* population. Specifically, fish that residualize in Whychus as redband therefore do not become steelhead and make no contribution to recovery of the ESU; and, the residualization of released fry has fundamentally changed the genetic structure of the Whychus Creek *O. mykiss* population.

Studies on life history plasticity in salmonids suggest the life history pathway exhibited by an individual fish is determined by the environmental context, and that the genetic threshold that cues one pathway or another varies as a function of local adaptation (Sogard et al 2012). In the Mokelumne River in California's Central Valley, *O. mykiss* growth rate as a function of time of emergence and potentially stream temperature and fish density resulted in a high proportion of *O. mykiss* adopting a resident life history, relative to *O. mykiss* in the American River that experienced warmer stream temperatures and potentially lower densities, and exhibited higher growth rates and uniform adoption of an anadromous life history (Sogard et al 2012). These results suggest that growth rate as a function of density, stream temperature, and food availability may be contributing to presumed high rates of *O. mykiss* residualization in Whychus Creek in contrast to Crooked River tributaries.

Life histories and restoration needs in Whychus

Redd counts, juvenile outmigration trapping, and radio tracking adult steelhead and Chinook salmon following their release upstream of the Round Butte dam are providing data that allow preliminary description of life history timing and associated locations of life history activities for resident redband and the two reintroduced species (Appendix A). This knowledge in turn suggests when and where it is most critical for restoration partners to focus efforts to restore sufficient conditions that will support each life history. Redband and steelhead rearing occur year-round in Whychus, requiring 18°C stream temperatures year-round. Redd counts conducted between 2007 and 2014 show the greatest concentration of redds consistently located in the Alder Springs reaches and at Rimrock Ranch. Steelhead spawning is anticipated to occur in April and May in Whychus with Chinook spawning from August to October, recommending flow management that will maintain temperatures below 13°C at Alder Springs and Rimrock Ranch during these months.

Conclusions

O. mykiss and spring Chinook study data from Whychus Creek and the Deschutes River Arm of Lake Billy Chinook cumulatively depict low juvenile abundance, low rates of spawning, and few adult returns. Survey data from 2014 are consistent with previous years' findings showing low *O. mykiss* densities and no statistically significant trend in *O. mykiss* abundance. 2014 redd count numbers were up from 2013 and 2012 but remained low relative to 2006-2011 counts. Two returning steelhead and two returning Chinook ascended the Deschutes from Lake Billy Chinook in 2014, none of which entered Whychus.

Genetic analyses show the 2013 *O. mykiss* adult population in Whychus Creek represented primarily Round Butte Hatchery stock, suggesting a high rate of residualization and a widespread genetic effect of reintroduction on the natural-origin *O. mykiss* population. PGE will conduct further analyses to differentiate between juvenile redband and steelhead five and ten years (in 2017 and 2022, respectively) after returning steelhead were first passed upstream of the dams in 2012. Additional genetic analyses will allow researchers to support preliminary findings and better understand the status and trends of resident redband and reintroduced steelhead populations in Whychus Creek and other Upper Deschutes Basin rivers and tributaries.

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

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

Acknowledgements

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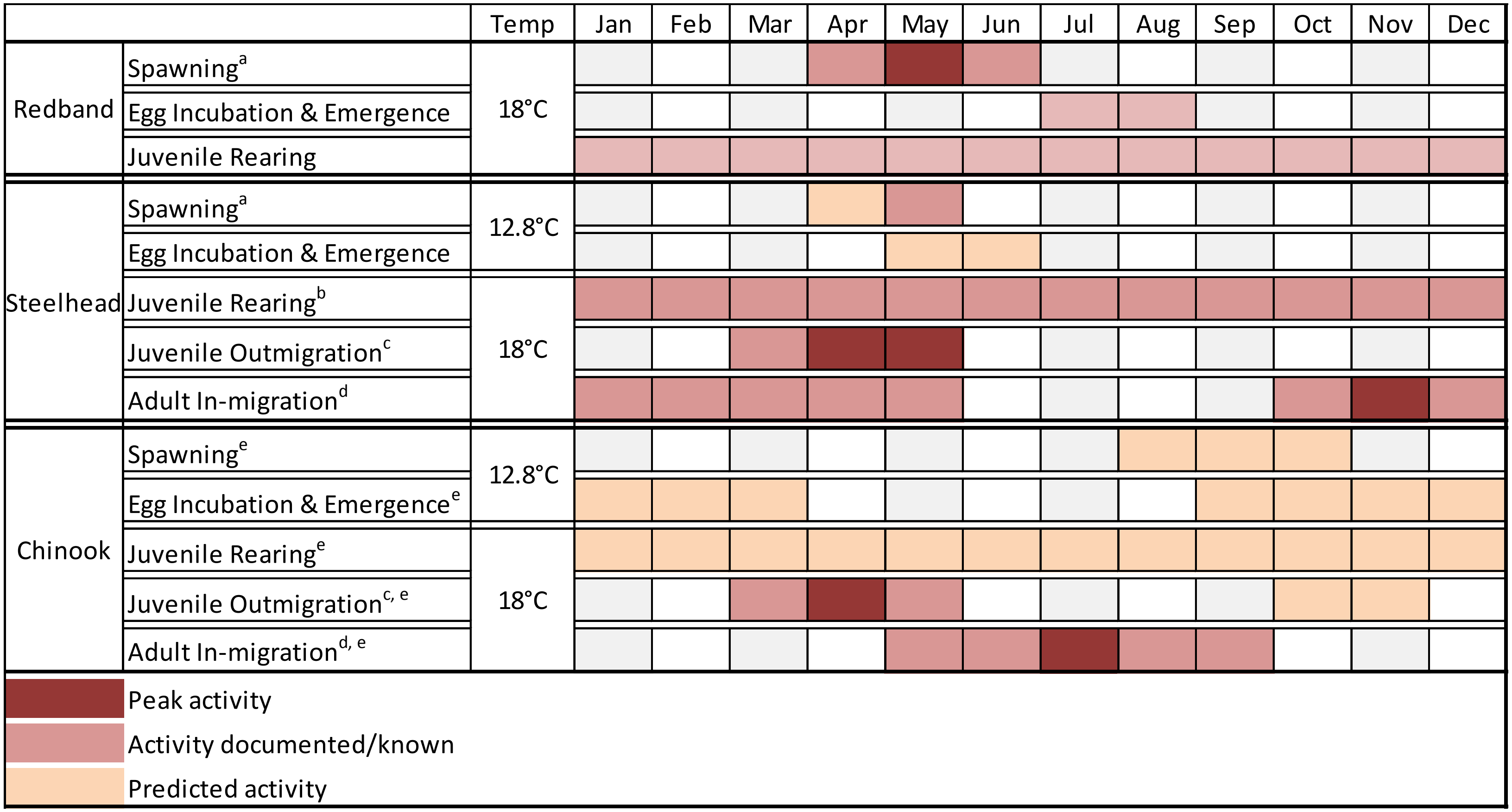
APPENDIX A Preliminary life history timing and temperature requirements of redband, steelhead, and Chinook salmon in Whychus Creek.



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