Upper Deschutes Watershed Council
Technical Report

2012 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
EPA	Environmental Protection Agency
FERC	Federal Energy Regulatory Commission
NIST	National Institute of Standards and Technology
NOAA	National Oceanic and Atmospheric Admistration
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
OWEB	Oregon Watershed Enhancement Board
OWRD	Oregon Water Resources Department
PGE	Portland General Electric
TSID	Three Sisters Irrigation District
UDWC	Upper Deschutes Watershed Council
EPA	United States Environmental Protection Agency
USFS	United States Forest Service
USGS	United States Geological Survey

0S	Age 0+ summer salmonid stage
0W	Age 0+ winter salmonid life stage
1S	Age 1+ summer salmonid life stage
1W	Age 1+ winter salmonid life stage
7DMAX	Seven day moving average maximum temperature
BACI	Before After Control Impact
°C	Degree Celsius
cfs	Cubic feet per second
CI	Confidence Interval
CL	Confidence Level
df	Degrees of freedom
DO	Dissolved oxygen
°F	Fahrenheit
mg/L	Milligrams per liter
OAR	Oregon Administrative Rules
QA/QC	Quality assurance / quality control
S	Standard distance from regression line
Spawning	Spawning and rearing salmonid life stages
StDev	Standard deviation from mean
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 and reintroduce anadromous fish above the Pelton Round Butte dam complex 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 to restore habitat conditions in Whychus Creek since the mid 1990s. 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 barrier 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's approach to monitoring restoration effectiveness in Whychus Creek acknowledges these limitations and seeks to leverage limited resources to improve monitoring. The UDWC has 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 the stream structures and functions or lead to responses in target species. Practitioners would then design each restoration activity as an experiment and evaluate their hypotheses using controls, statistical tools and other standard experimental practices.

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

The monitoring approach selected by the UDWC focuses on tracking the status and trends of key physical and biological indicators in Whychus Creek. The UDWC selected these indicators based on a conceptual model of factors limiting salmonid production in the creek (Figure 1). They expect that ongoing restoration actions will ameliorate the limiting factors identified in the conceptual model and that selected indicators will respond to changes in these limiting factors. This approach will not test cause and effect relationships between restoration actions and changes in selected indicators. It will demonstrate whether these indicators have moved closer to desired conditions. The UDWC drew indicators from seven broad categories: stream flow, water quality, habitat quality, stream connectivity, fish entrainment, macroinvertebrates, and fish populations. Each chapter of the 2012 Whychus Creek Monitoring Report assesses indicators in one of these categories.

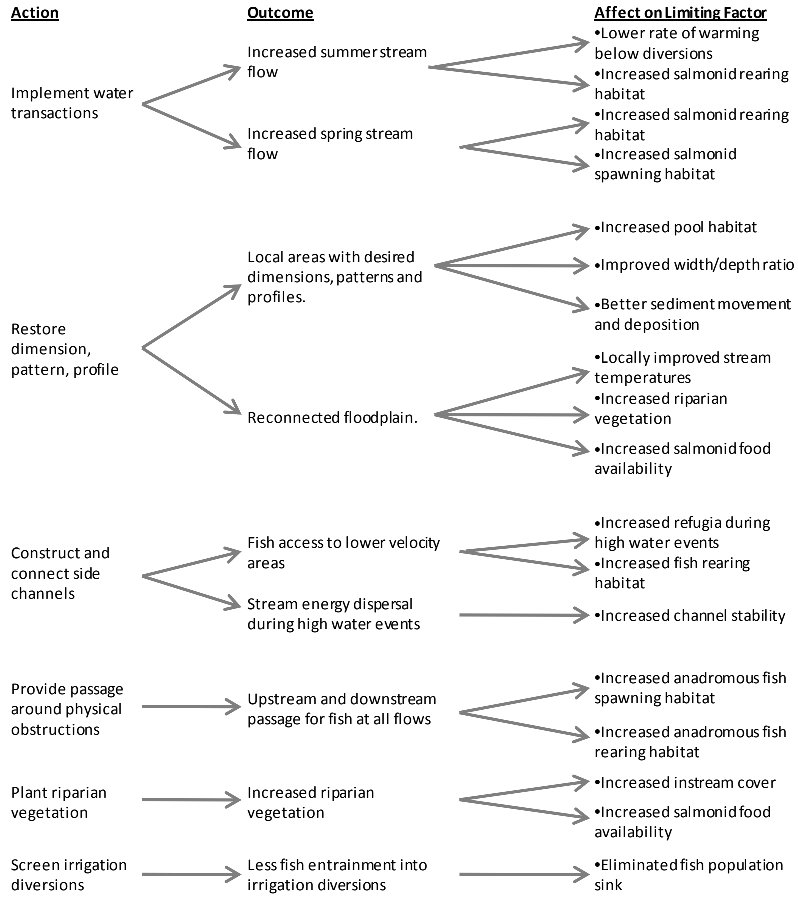

Figure 1.
This conceptual model illustrates the expected influences on each life stage of resident and anadromous salmonids in Whychus Creek. The UDWC expects that the ongoing restoration actions will affect the limiting factors identified in the conceptual model.

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.

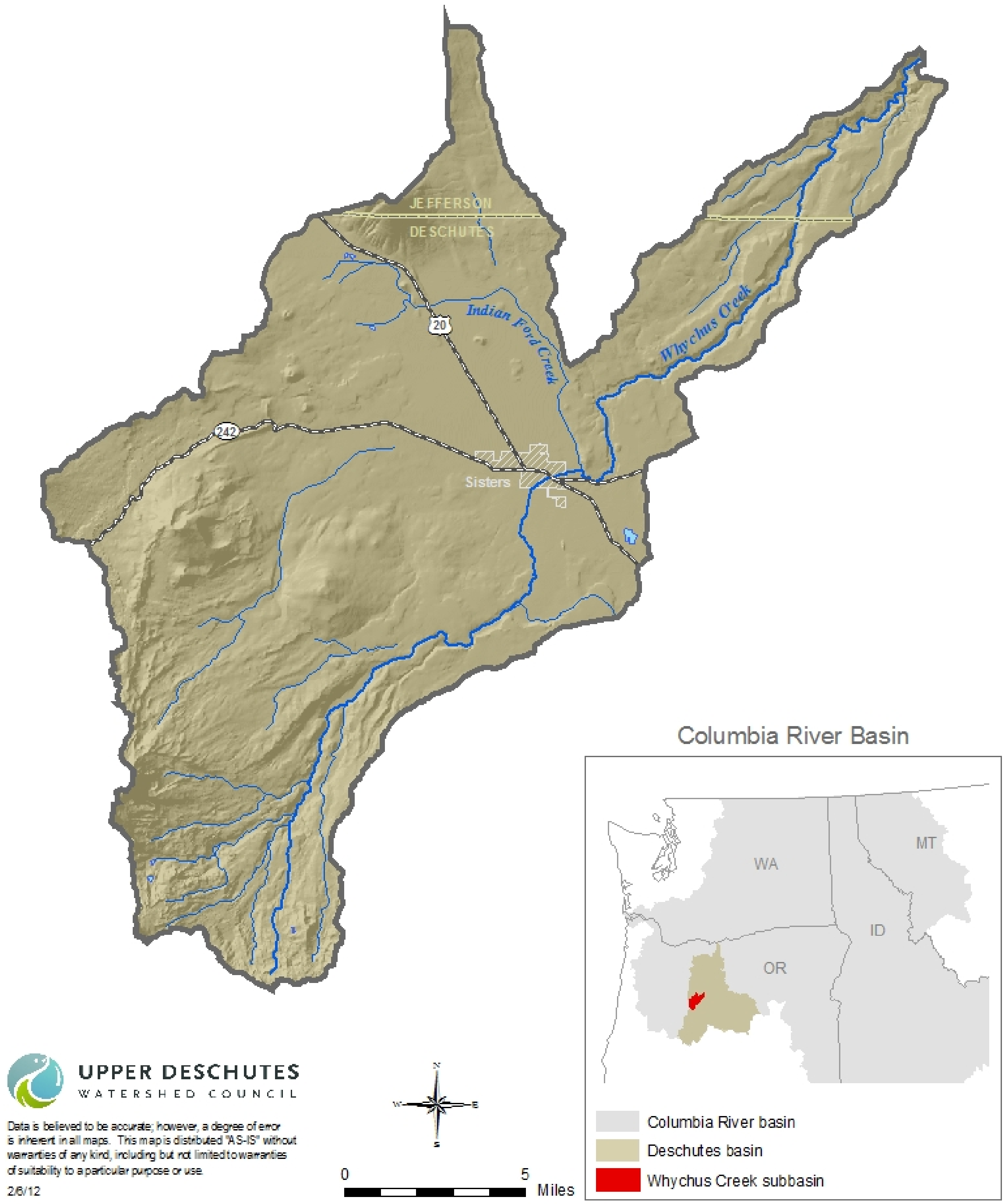

Figure 2.
Whychus Creek extends from the Cascade Range to the Deschutes River. The creek's watershed encompasses approximately 162,000 acres of Deschutes and Jefferson Counties in central Oregon.

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 upon 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 25 (UDWC 2008). 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 and habitat restoration upstream of the Pelton Round Butte complex. 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 envision the implementation of these projects over a ten-year period beginning in 2009.

Technical Studies

Annual technical studies analyze and interpret available data to examine the status and trends of physical and biological indicators in Whychus Creek. These studies document changes from baseline conditions following the implementation of large scale habitat restoration actions along the creek, including streamflow restoration, channel realignment, fish passage improvements, screening of irrigation diversions, and other restoration actions. Baseline conditions are reported in the 2009 Whychus Creek Monitoring Technical Report (UDWC). These conditions were inventoried following some streamflow restoration but prior to any other restoration efforts.

Golden (2013) documents summer stream flow conditions in Whychus Creek from 2000-2012. It focuses on metrics representing low flow conditions in the creek. Mork (2013a) answers questions related to stream temperature in Whychus Creek. It draws on thirteen years of data to examine water quality in relation to state standards and to stream flow restoration. Restored stream flow has affected metrics in each of these reports.

Two reports quantify habitat improvements resulting from restoration projects completed subsequent to baseline analyses. Mork (2013b) documents the status of fish passage barriers as a measure of stream connectivity along the creek. Restoration partners expect to provide passage at each of the original six barriers identified in the 2009 report. Mork (2013c) discusses reductions in fish entrainment potential on the creek. It sets unscreened irrigation diversions and the cumulative flows diverted through them as a proxy for entrainment potential with the expectation that restoration partners will screen each of these diversions in the future. No new habitat surveys were conducted on Whychus Creek in 2012; accordingly this report does not include an updated habitat quality study.

Two additional reports update the status of biological conditions in the creek. Mork (2013d) summarizes PGE's 2012 steelhead and chinook survey results. It outlines the status of fish populations in the creek and discusses how additional sampling and new methods planned for future years will expand the current understanding of these populations. Although the ongoing reintroduction of steelhead and chinook in Whychus Creek prevents meaningful application of population data for these species as an indicator of restoration effectiveness, tracking fish population trends in Whychus provides essential information for future evaluation of reintroduction and restoration efforts. Mazzacano (2012) examines four years of macroinvertebrate data to identify trends in macroinvertebrate community composition before and following extensive streamflow restoration.

These six reports evaluate improvements in stream conditions in 2011 as measured by the status of physical and biological indicators subsequent to major streamflow and channel restoration and irrigation diversion retrofits. The reports and the data that they contain will help restoration partners to understand the effectiveness of their actions at moving the creek toward desired conditions. Restoration partners expect 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. Annual minimum 30 day moving average flows increased or remained constant in every year except for 2005 and 2009. May median flows exhibited both inter-annual and intra-annual variation. May median flow ranged from a low of 5.4 cfs in 2003 to a high of 64 cfs in 2012. August median flows also exhibited inter-annual and intra-annual variation but intra-annual variation was typically lower than in May. August median flow ranged from a low of 2.6 cfs in 2002 to high of 32 cfs in 2011. These results suggest that Whychus Creek still experiences low flows during both late spring/early summer and late summer/early fall flow, two periods when irrigation demands generally exceed water availability. Extreme flows, however, appear to be decreasing in magnitude during both of these periods. These results highlight the need to understand whether low flows during these periods limit ecosystem function and, if so, to focus on restoration efforts during these periods. As restoration continues to increase flows in Whychus Creek, restoration partners should continue to evaluate both early and late season flow as well as extreme low flows to fully describe restoration outcomes.

Introduction

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

The entire hydrograph affects what a stream looks like and how it functions (Poff *et al* 1997). Different components of the hydrograph may drive different ecological processes (Doyle *et al* 2005). Changes in stream flow can affect biological characteristics such as macroinvertebrate assemblages (Dewson *et al* 2007, Konrad *et al* 2008, James *et al* 2008, Monk *et al* 2006, Wills *et al* 2006), fish communities (Xenopoulos *et al* 2006, Decker e*t al* 2008), and riparian vegetation (Stromberg *et al* 2005). By removing up to 90% of the stream flow from Whychus Creek, irrigation diversions have eliminated all but the low flow components of the hydrograph during the summer and likely affected each of these characteristics. Monitoring the status and trends of stream flow in Whychus Creek will illuminate whether the stream is moving towards or away from desired conditions.

Hydrologists have developed a wide range of hydrograph related metrics to track stream flow conditions over time. These different metrics relate to different components of the hydrograph that affect physical and biological conditions in a stream. Olden and Poff (2003) identify 171 metrics that appeared in 13 papers. These metrics relate to the magnitude, frequency, rate of change, duration, or timing of flow events. Monk *et al* (2006) 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* 2006, Yang *et al* 2008, Gao *et al* 2009). Researchers have not yet identified a single subset of metrics that represent alteration in all types of streams. Different types of streams have different hydrologic characteristics. For example, groundwater dominated streams exhibit relatively low seasonal variability while snowmelt dominated streams exhibit clear seasonal patterns. The type of stream, surrounding geography, and the desired conditions in that stream define the appropriate set of metrics.

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

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

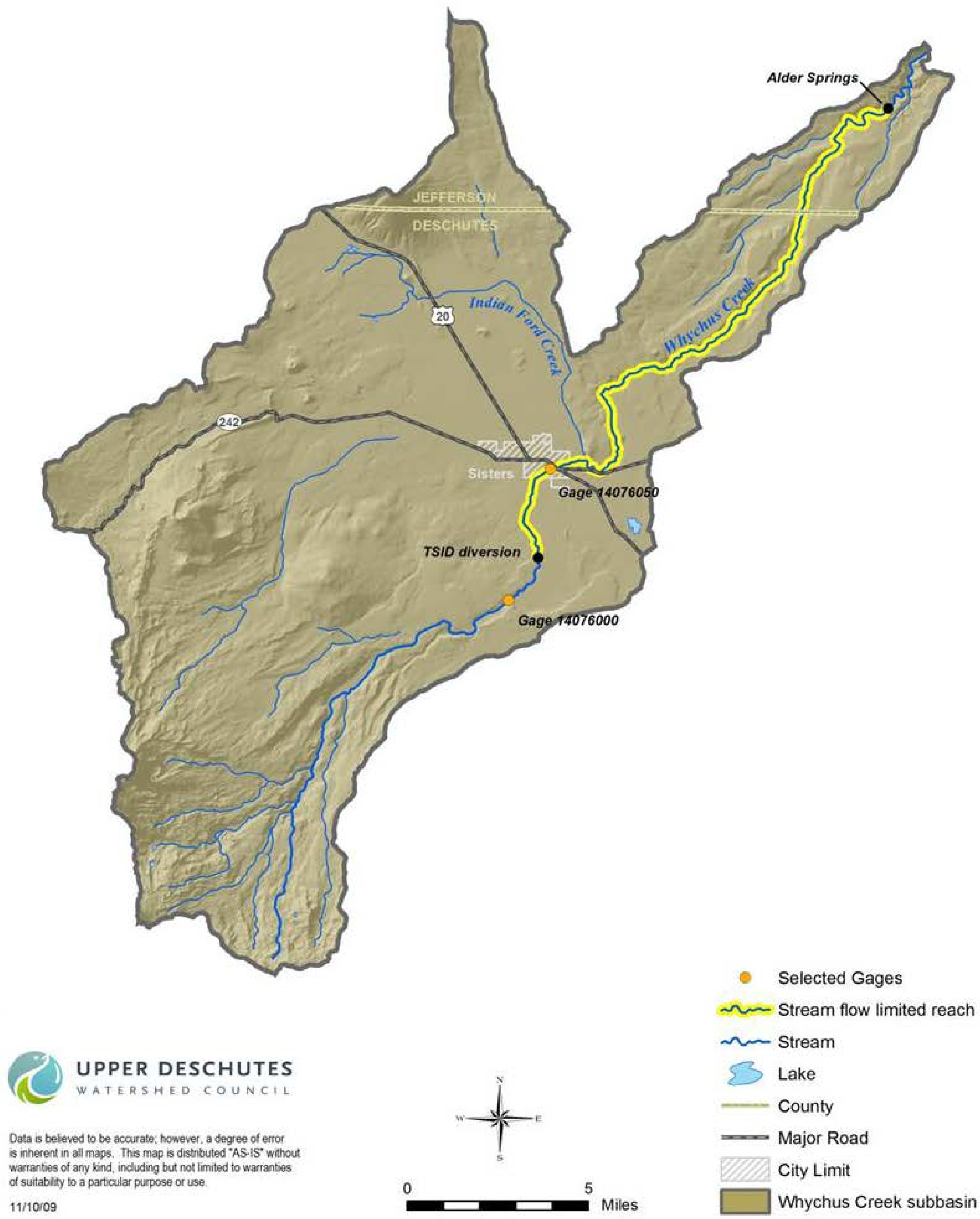

Figure 1.
Stream flow limits stream function in Whychus Creek downstream from the Three Sisters Irrigation District Diversion. Spring inputs near the mouth of Whychus Creek increase stream flow and improve conditions in the creek.

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

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

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

Minimum 30 Day

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

May Median

May median flow may provide a general indicator of spawning habitat availability in Whychus Creek. Redband trout spawning in the Deschutes Basin centers on the month of May (ODFW 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 2012.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 2012. 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 2012 reflects conditions in the creek during ongoing restoration efforts.

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

May Median

The DRC used spreadsheet software to determine the median daily average flow during the month of May for years 2001 through 2012. 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 2012. 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 16.00 cfs in 2008. It increased or remained constant each year except for 2005 and 2009.

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

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

May Median

The DRC analyzed stream flow data as described above. Average May flow in Whychus Creek at the Oregon Water Resources Department's gage number 14076050 exhibited both inter-annual and intra-annual variation (Figure 2). Median flow during the month of May ranged from a low of 5.4 cfs in 2003 to a high of 64.0 cfs in 2012. 2006 exhibited the greatest intra-annual variation in May flow, with a 20th percentile value of 22.0 cfs and an 80th percentile value of 122.0 cfs.

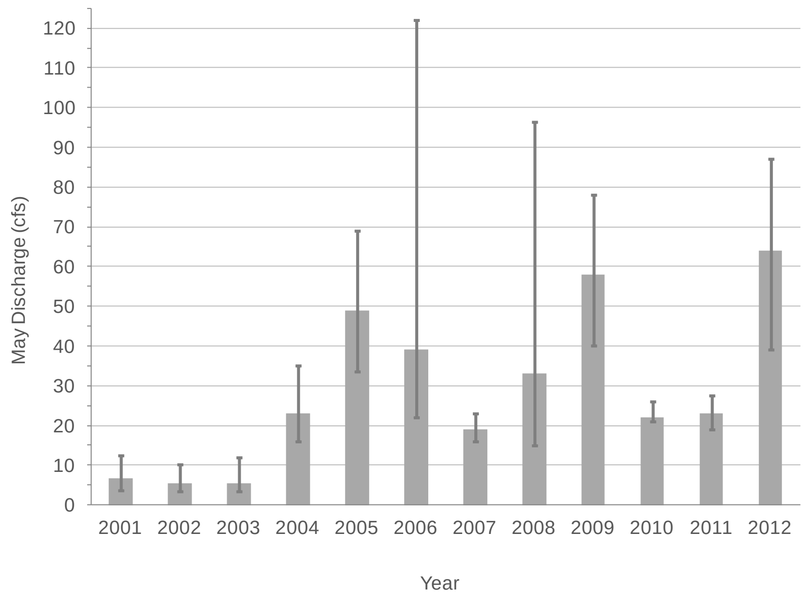

Figure 2.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of May provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of May at this location.

August Median

Median discharge during the month of August exhibited both inter-annual and intra-annual variation at gage number 14076050 (Figure 3). 2002 exhibited the lowest median discharge during the month of August, with a median daily average discharge of 2.6 cfs. 2011 exhibited the highest median discharge during the month of August, with a median daily average discharge of 32.0 cfs. 2008 exhibited the greatest intra-annual variation in discharge, with a 20th percentile discharge of 27.5 cfs and an 80th percentile discharge of 45.0 cfs.

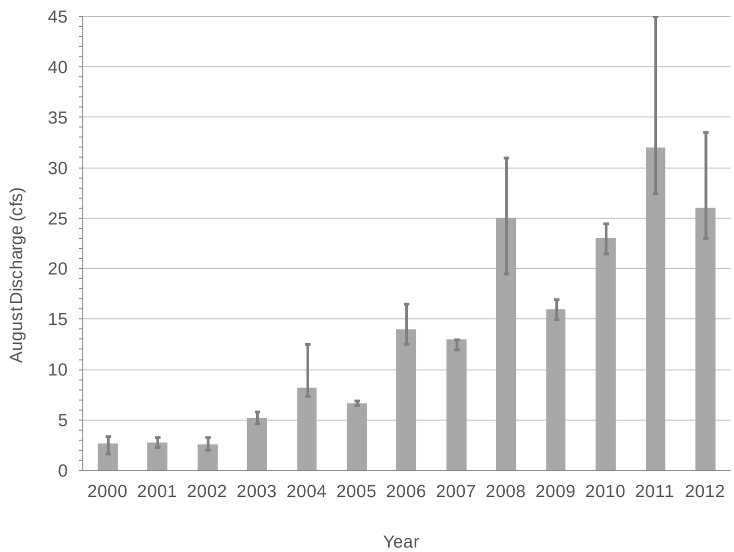

Figure 3.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of August provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of August at this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

Discussion

The analyses in this report describe baseline stream flow conditions in Whychus Creek at the beginning of a ten-year period of intensive restoration. They focus on the period from 2000 through 2012. 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 six out of twelve years (OWRD 1996a, 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 1996b, Figure 3).

Restoration partners have focused on late summer stream flow as a metric for restoration effectiveness. Late spring/early summer stream flow may also be important for stream function. As noted earlier, redband trout spawning centers on the month of May (ODFW 2005). Consistently low stream flow 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 are improving. 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 only in 2008 and 2010 (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. 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 beyond base flow. High stream flow events continue to occur before, during and after the irrigation season. Irrigation operations have reduced the magnitude of but not eliminated these events. Although describing a desired hydrograph would better inform restoration partners about the status of the stream flows in Whychus Creek, it would likely not improve the ability of restoration partners to address other hydrograph components. The absence of any storage reservoirs along Whychus Creek and the continued presence of high flow events have reduced the priority of evaluating non-base flow components of the hydrograph.

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

Acknowledgements

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

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

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Abstract

Diversion of almost 90% of summer streamflow and channelization of over 50% of the length of Whychus Creek have degraded water quality, leading to an ODEQ listing of water quality limited in 2002, 2004, and 2010. The Upper Deschutes Watershed Council monitored temperature from 1995 through 2012 at eleven sites representing diverse flow conditions in Whychus Creek. This report incorporates 2012 data to 1) evaluate the current status of temperature in Whychus Creek in relation to state standards for salmonid spawning, rearing and migration; 2) quantify changes in temperature in reaches with restored flows; and 3) refine target flows projected to produce temperatures that meet state standards. Temperatures exceeded state rearing and migration standards at four monitoring sites in 2012 for a total of 25 days, up from three sites and 19 days in 2011 but lower than the 49 days during which temperatures exceeded the standard across five sites in 2010. Temperatures never exceeded the lethal threshold for salmon and steelhead in 2012, for the third year in a row. Regression of 1995-2012 temperature and flow data identified 75 cfs as the minimum flow necessary to meet the 18° temperature standard at FS Road 6360, within the range of the 66-78 cfs predicted by 2008, 2010 and 2011 models. Our results show that streamflow restoration has already improved temperature conditions for re-introduced salmon and trout, and advance our understanding of temperature and flow on Whychus Creek to better inform future watershed restoration efforts.

Introduction

Restoration partners have identified the Whychus Creek watershed as a priority watershed for conservation and restoration within the upper Deschutes Basin (NWPPC 2004, UDWC 2006). Diversion of almost 90% of average summer flows and historic channelization of nearly 50% of the creek length have created conditions that contribute to elevated stream temperatures and may compromise other water quality parameters. In 1998, 2002, 2004, and 2010, Whychus Creek was listed by ODEQ under Clean Water Act Section 303(d) as water quality limited with TMDLs needed for temperature (Category 5) and categorized as having insufficient data for assessment for dissolved oxygen and pH (Category 3) (Table 1, Figure 1). The 2012 Integrated Report is pending (ODEQ 2013).

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

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

ODEQ state temperature standards have been established to protect fish and other aquatic life in Oregon waterways (ODEQ 2009). The year-round temperature standard applied to Whychus Creek for salmon and trout rearing and migration specifies that seven-day moving average maximum (7DMAX) temperatures are not to exceed 18˚C. The 2002 303d list also identified Whychus Creek as not meeting the 12.8˚C state temperature standard for salmon and steelhead spawning. Neither the 2004 nor the 2010 303(d) lists applied this criterion to Whychus Creek because anadromous fish were not spawning in Whychus Creek when data for these lists were collected. However, this habitat use may resume, and the spawning temperature standard become relevant, as salmon and steelhead reintroduced in 2007 begin to return to the creek. Steelhead spawning season has yet to be identified for Whychus Creek. This report references the January 1 – May 15 spawning season identified for the Lower Deschutes sub-basin. The State of Oregon 1992-1994 Water Quality Standards Review (ODEQ 1995) identified 24˚C as the lethal temperature threshold for salmon and trout.

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

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

The 2004 and 2010 ODEQ 303(d) lists classify Whychus Creek as having insufficient data for assessment for dissolved oxygen and pH. UDWC analyses of data collected from 2006 to 2008 indicated that Whychus Creek met state dissolved oxygen standards for steelhead and salmon rearing and migration, although dissolved oxygen levels did not consistently meet state criteria for salmon and trout spawning (Jones 2010). Because dissolved oxygen saturation is directly affected by temperature, we can expect dissolved oxygen levels to track temperature trends. While observed trends in stream temperature continue to demonstrate cooling, and in the absence of other novel environmental conditions, we expect dissolved oxygen levels to improve or remain constant. Under these circumstances temperature data are an appropriate proxy for dissolved oxygen data, and indicate dissolved oxygen levels that will continue to meet the state standard for salmon and trout rearing and migration. UDWC discontinued monitoring dissolved oxygen on Whychus Creek in 2009 on this premise. A consistent warming trend in temperature would flag potentially deteriorating dissolved oxygen conditions and warrant resuming monitoring of dissolved oxygen. Although 2006-2008 data indicated pH standards were not consistently met in the summer, low pH values were attributed to the influence of volcanic soils and were not expected either to limit ecological function or to be affected by increased flows with streamflow restoration. Accordingly this parameter was also discontinued as of 2009. While this report does not present dissolved oxygen or pH data, we consider the observed trends in temperature to provide a strong surrogate measure of water quality in Whychus Creek. For further discussion of temperature, dissolved oxygen, pH, and state standards for each parameter, refer to *Whychus Creek Water Quality Status, Temperature Trends, and Streamflow Restoration Targets* (Jones 2010).

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

Whychus Creek Water Quality Status, Temperature Trends, and Streamflow Restoration Targets (Jones 2010) presented baseline information on the status of temperature, dissolved oxygen, and pH in Whychus Creek, demonstrated temperature response to increased flows from streamflow restoration in three downstream reaches, and developed a regression model describing the relationship between temperature and flow to identify streamflow restoration targets. Subsequent versions of this report incorporated 2009, 2010 and 2011 data to update 2006-2008 analyses (Mork 2012, Mork and Houston 2013). We present revised analyses including 2000-2012 temperature and flow data to evaluate the 2012 status of temperature in Whychus Creek in relation to state standards for salmonid spawning, rearing and migration, quantify changes in temperature in reaches with restored flows, and refine flow targets projected to produce temperatures that meet state standards.

Table 1. 2010 Oregon Clean Water Act Section 303(d) status of Whychus Creek.

	Parameter	**Temperature**		**Dissolved Oxygen**		**pH**	
	Beneficial Use	Salmon & Trout Rearing & Migration	Steelhead Spawning	Salmon & Steelhead Non-Spawning	Salmon & Trout Spawning	Multiple Uses	Multiple Uses
	Season	Year Round	January 1 - May 15	Year Round	January 1 - May 15	Fall/ Winter/ Spring	Summer
	Standard	18° C	12° C	8.0 mg / L @ 90% Sat	11.0 mg / L @ 90% Sat	6.5-8.5 SU	6.5-8.5 SU
ODEQ Reach	0 - 40.3	TMDL Needed	Not Applicable	Not Applicable	Insufficient Data for Section 303(d) Assessment	Insufficient Data for Section 303(d) Assessment	Insufficient Data for Section 303(d) Assessment
	1 - 13.3	Not Applicable	Not Applicable	Insufficient Data for Section 303(d) Assessment	Not Applicable	Not Applicable	Not Applicable
	13.3 - 40.3	Not Applicable	Not Applicable	Insufficient Data for Section 303(d) Assessment	Not Applicable	Not Applicable	Not Applicable

Source: ODEQ 2011

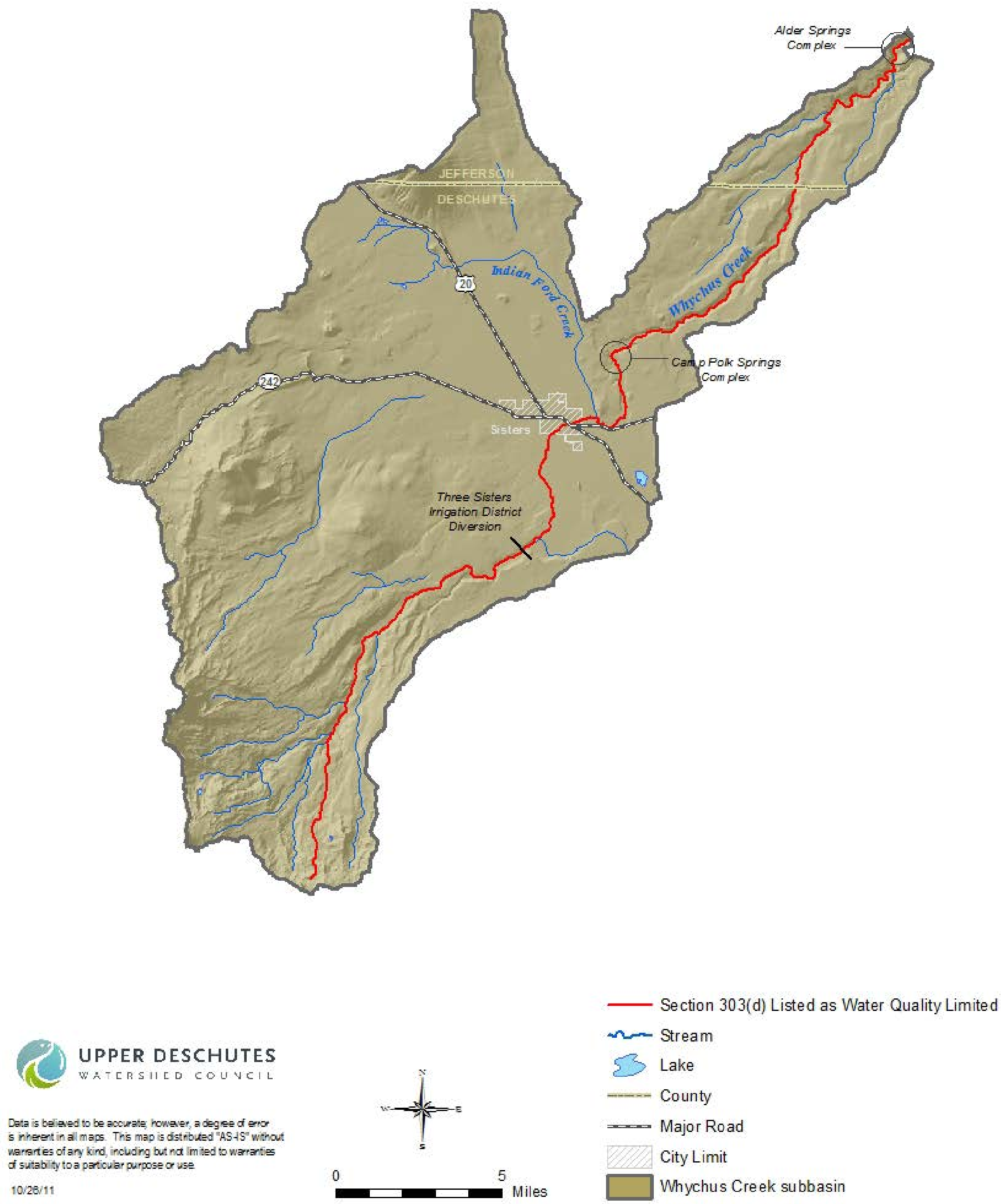


Figure 1.
Whychus Creek is listed as Water Quality Limited from river mile (RM) 0.0 to RM 40.3 under ODEQ's 2002, 2004, and 2010 303(d) lists. (ODEQ 2011).

Methods

Data collection

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

In 2009 UDWC, Deschutes Land Trust (DLT), private landowners and other restoration partners reached an agreement to restore 1.9 miles of the historic channel of Whychus Creek at Rimrock Ranch. The planned restoration will divert the creek from the existing channel into the historic meadow channel, and the UDWC monitoring station historically located on the existing channel will no longer be creekside. To replace this monitoring location and generate pre-restoration data above and below the restoration project site UDWC established two new temperature monitoring stations, one upstream and one downstream of the planned restoration. As of 2009 UDWC discontinued temperature monitoring at the old Rimrock temperature monitoring station at WC 009.00 and began monitoring temperatures at these two locations.

We obtained average daily streamflow (QD) data for Whychus Creek from Oregon Water Resources Department (OWRD) gage 14076050 at the City of Sisters (OWRD 2013). This gage is located downstream from the Three Sisters Irrigation District diversion and other major irrigation diversions. We used data collected at this gage from 2000 to 2012 in this report, including some data considered by OWRD to be provisional and subject to change.

Temperature status

We evaluated 2012 seven day moving average maximum daily temperatures (7DMAX) in relation to the 18˚C state temperature standard for salmonid rearing and migration and the 12.8˚C state standard for salmonid spawning according to methods described in the ODEQ *Assessment Methodology for Oregon's 2004/2006 Integrated Report on Water Quality Status* (ODEQ 2006). At monitoring sites where July 7DMAX temperatures exceeded the18˚C standard, we compared temperatures to flows recorded at Sisters City Park in relation to ODFW water rights and DRC streamflow restoration targets. We also compared the status of Whychus Creek temperatures in 2012 to 2000-2011 results.

We calculated the average rate of temperature change for Whychus Creek on the hottest water day of 2012 from the farthest upstream site at RM 38 (WC 038.00) to the farthest downstream site at RM 0.25 (WC 000.25) by dividing the difference in temperature between these two sites by the distance between the two sites. We defined the hottest water day as the day with the single hottest seven day moving average maximum temperature (7DMAX). For each reach between temperature monitoring sites we calculated the amount by which the longitudinal rate of change was higher or lower than the average rate of change. Higher than average longitudinal changes in temperature identify reaches in which the rate of warming increased, allowing restoration partners to prioritize these areas for research and restoration planning. Lower than average longitudinal changes in temperature highlight reaches where cooling occurred and which may accordingly be prioritized for additional conservation measures.

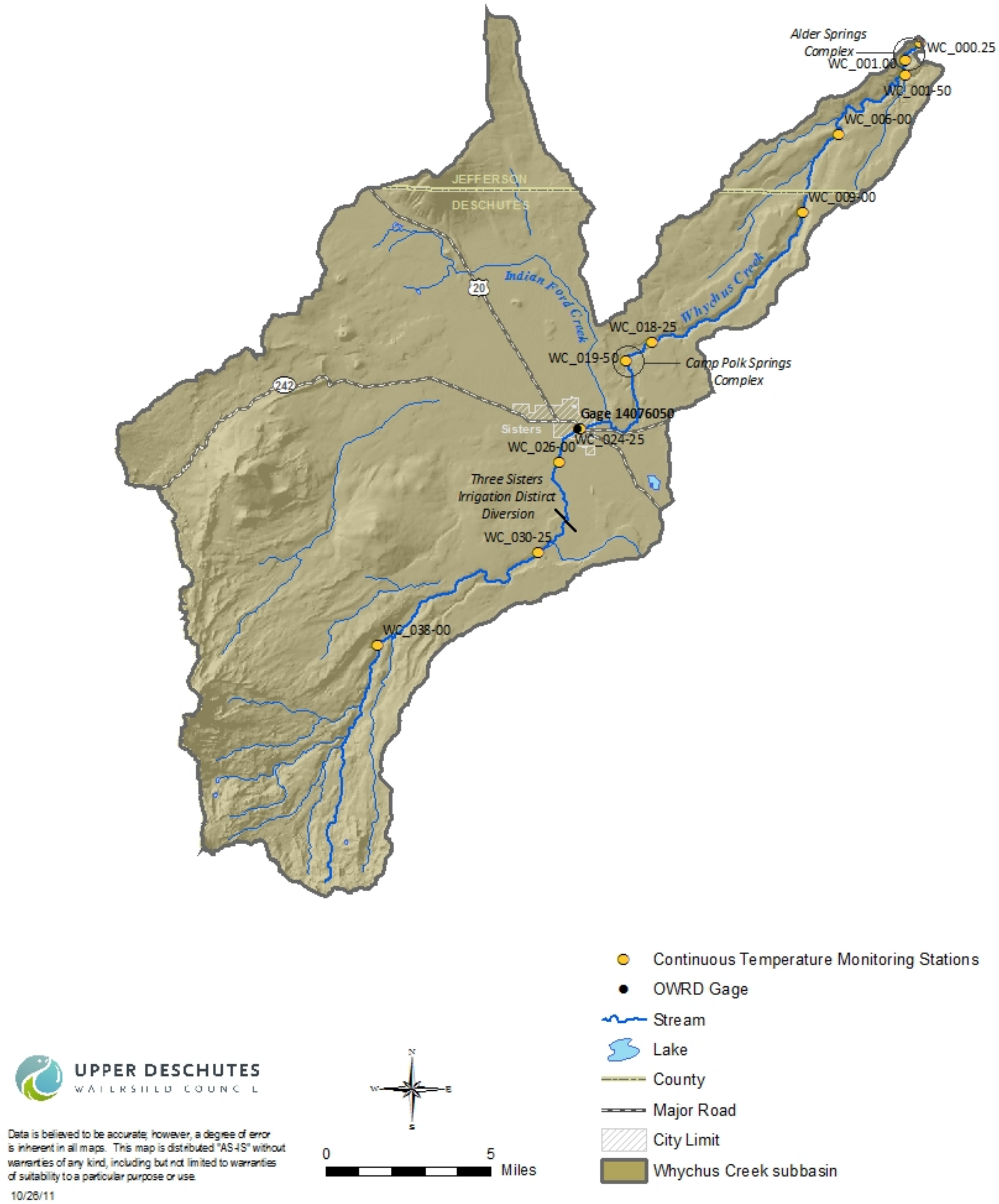


Figure 2.
Continuous temperature monitoring stations monitored in 2012, and OWRD Gage 14076050, on Whychus Creek.

Target Streamflow

We added 2012 data to the 2000-2011 dataset to perform a temperature-streamflow regression that refines the target streamflow projected to result in temperatures at or below the 18˚C state standard. We used July 7DMAX temperature data for each year included in the analysis from two monitoring stations, WC 024.25 and WC 006.00, with streamflow data from the OWRD gage at Sisters, to identify the streamflow required at each of these sites to achieve a 7DMAX temperature of 18˚C. Temperature data from WC 024.25 represent stream conditions immediately below major irrigation diversions; data from WC 006.00 represent the historical worst temperature conditions on the creek, and thus the location that is both most critically in need of and also stands to benefit the most from streamflow restoration.

We restricted data included in the regression to one month of the year to reduce the effect of inter-annual seasonal variation in the analysis (Helsel and Hirsch 2002) and identified July as the month during which the hottest water day has historically occurred most frequently (UDWC 2003; UDWC 2008c). Daily streamflow data for all July days from 2000-2012 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 2013).

To estimate temperatures at corresponding flows for the two locations we performed a regression of temperature and flow data. The resulting equations accurately represent the relationship between flow and temperature and can be used to calculate temperature values for the specified locations, within the evaluated time period, and within the range of flows observed. We paired 7DMAX temperature with the corresponding natural log of the average daily flow (LnQD) for each July day included in the analysis, then ranked flow values, assigned all July temperature records to their corresponding flow value, and calculated the mean of all 2000-2012 July 7DMAX temperatures observed at each flow level (Mork and Houston 2013). The seven day moving average maximum temperature for a given day is the average of the maximum temperature for that day, the three days prior, and the three days following; we paired the 7DMAX for a given day with the flow for the same day to best match the 7DMAX temperature to flow conditions on both the first and seventh days represented by the 7DMAX temperature. While this approach does not reflect flows corresponding to maximum daily temperatures on the fifth, sixth, or seventh days included in the 7DMAX temperature, we selected the flow corresponding to the 7DMAX for the same date as potentially more representative of the flow on both the first and seventh days included in the 7DMAX than if we used the flow from the seventh day to correspond to the 7DMAX temperature and thereby to represent the flow corresponding to the temperature from the first day of the 7DMAX.

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

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

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

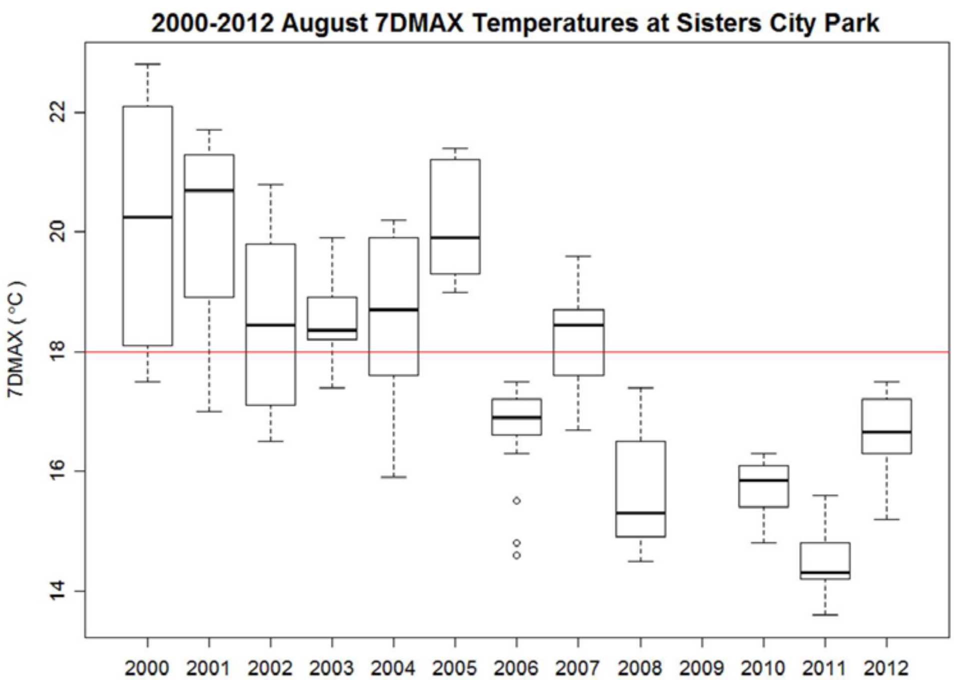
We compared the resulting 2000-2012 temperature-flow regressions and predicted temperatures at given flows for each site to Heat Source model scenarios for the same locations on Whychus Creek (Watershed Sciences and MaxDepth Aquatics 2008). Because 33 cfs is the DRC streamflow restoration target, and because available Heat Source scenarios assume 33 cfs at WC 024.25 and 62 cfs at WC 006.00, we compared 2000-2012 temperature calculations for these flows.

Results

Temperature status

Although the overall temperature trend shows seven-day moving average maximum (7DMAX) temperatures decreasing from 2000-2012 (Figure 3), 7DMAX temperatures were warmer in 2012 than in 2011. Temperatures at Sister's City Park, three miles downstream of the most significant irrigation diversion on the creek, remained below the 18˚C standard in 2012 for the fourth year in a row, with only one year on record exceeding the state standard since 2006 (Figure 3); 7DMAX temperatures exceeded the 18˚C standard at this site in every year for which data were available prior to 2006. Temperatures exceeded the 18˚C state standard for steelhead and salmon rearing and migration at four downstream sites in 2012, compared to three sites in 2011, but fewer than in 2010 (five sites) or 2009 (six sites) (Figure 4). (However, data were not available in 2012 for WC 001.50, where temperatures have exceeded 18˚C every year since 2008.) At Road 6360 (WC 006.00), historically the most impaired site for which temperature data are available, 7DMAX temperatures exceeded the 18˚C standard at flows of 21 to 51 cfs for 25 days in 2012, six more than in 2011 but still substantially lower than the 49 days in 2010 and 47 days in 2009 during which temperatures exceeded the standard at this site. Temperatures in excess of the 18˚C standard occurred from July 31 to August 24, 2012. Lethal temperatures were not recorded at any site on Whychus Creek in 2012, for the third year in a row.

Continuous temperature data were available for the 2012 spawning season from April 27 to May 15 for Road 6360 and from May 1-15 for the six sites upstream of Road 6360 (Figure 4). Temperatures recorded for these dates exceeded the 12.8˚C spawning habitat requirement and potential state standard for eight days representing 42% to 53% of spawning season days for which data were available, at four sites between WC 006.00 and WC 018.25 in 2012, down from five sites between WC 001.50 and WC 018.25 in 2011 and six sites between WC 001.50 and WC 019.50 in 2010. Data were not available for 2012 spawning season dates for any sites downstream of WC 006.00.



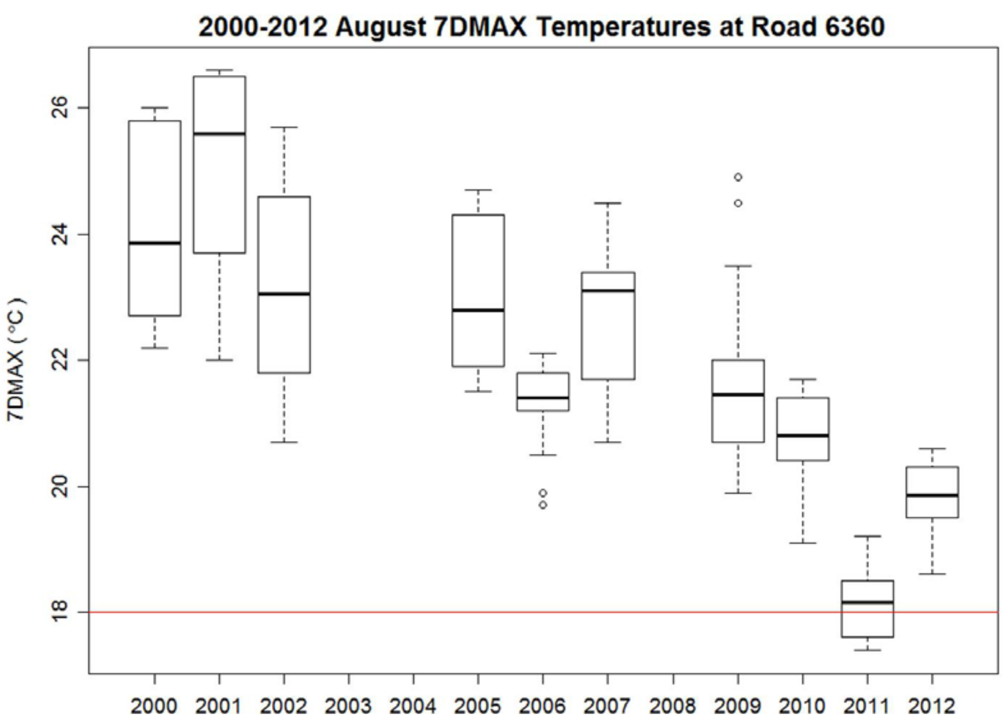


Figure 3.

August mean 7DMAX temperatures at a) Sisters City Park (WC 024.25) and b) Road 6360 (WC 006.00) chart a decreasing trend since 2000. Temperatures at Sisters City Park have exceeded the 18˚C state standard in only one year for which data are available since 2006; no data are available for this site for 2009. Temperatures at Road 6360 continue to exceed the 18˚C standard but have fallen dramatically since 2000.

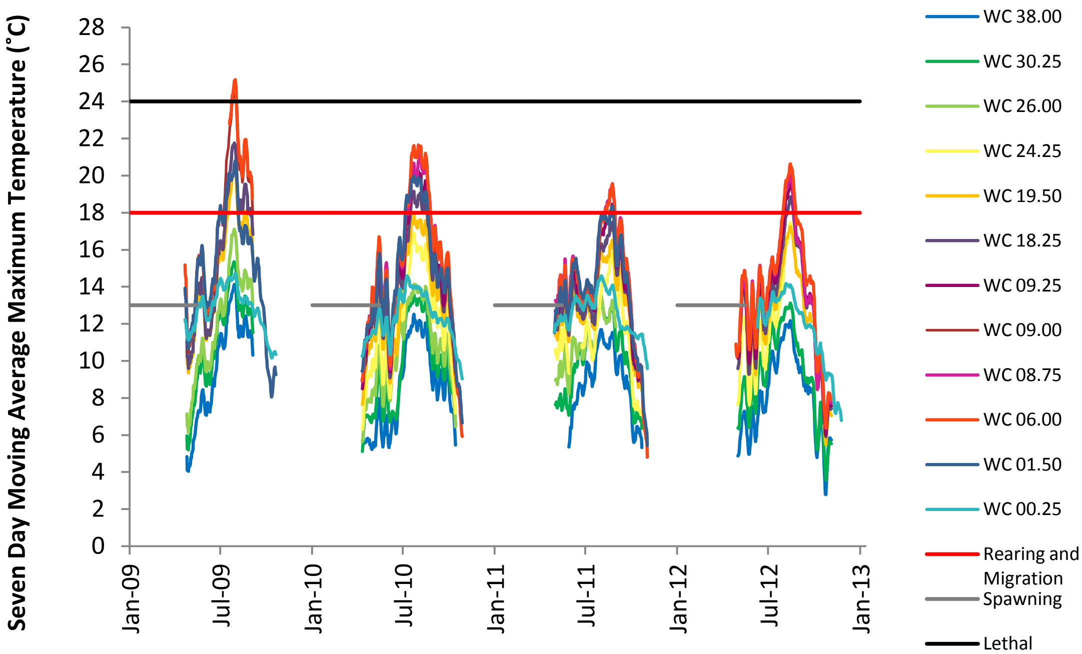


Figure 4.
Temperatures exceeded the 18˚C state standard for salmon and trout rearing and migration at four monitoring locations in 2012, one more than in 2011 but down from five sites in 2010 and six sites in 2009. Temperatures exceeded the January 1- May 15 12.8˚C potential spawning standard at four sites in 2012, compared to five sites in 2009 and 2011 and six sites in 2010. Temperatures exceeded the 24˚C lethal threshold at two sites in 2009, but have not met or exceeded the lethal threshold at any monitoring location along Whychus Creek since 2009.

The hottest water day in 2012 occurred on August 14th. Temperatures on the hottest water day at all sites monitored in 2012 were warmer than temperatures recorded on the hottest water day in 2011, but remained cooler at most sites than temperatures on the hottest water days from 2007-2010, with the exception of site WC 038.00, where temperatures on the hottest day in 2012 exceeded those in 2011 and 2010; WC 024.25, where 2012 hottest day temperatures exceeded those in all years shown other than 2007; and WC 000.25, where the 2012 temperature was the same as in 2011 (Figure 5). The average longitudinal rate of change in 2012 was 0.05˚C per mile, equivalent to the 2010 average, slightly lower than the 2011 average rate of 0.7˚C per mile, and higher than the 2008 and 2009 average rates of change (0.03˚C and 0.02˚C per mile, respectively) (Figure 6). The relatively higher average longitudinal rates of change for 2010 through 2012 reflect lower temperatures at the upstream-most site, with temperatures at the downstream-most site lower than in years with lower average longitudinal rates of change. Sites of above-average warming and cooling were similar to previous years' results. Whereas the greatest increase over the average rate of change in 2007, 2008, 2010 and 2011 occurred between WC 026.00 and WC 024.25, downstream of the TSID diversion and upstream of Sisters, in 2012 the greatest increase over the average rate of change occurred along the reconstructed channel at Camp Polk, from WC 019.50 to WC 018.25 (data from WC 024.25 are not available for 2009). Increased warming in this reach may reflect increased solar insolation along the new Camp Polk channel, in the absence of the mature riparian canopy that shaded the straightened channel in previous years. High rates of warming also continue to occur at the downstream monitoring stations for Rimrock Ranch, a channelized site characterized by high solar insolation which has been prioritized for stream channel restoration. The -1.1˚C rate of temperature change from Rd 6360 to Alder Springs was substantially lower than the average rate of change, reflecting the cooling effect of springs complex flows, but far less dramatically so than in 2009 (-4.9˚C), 2010 (-4.7˚C) and even 2011 (-3.4˚C).

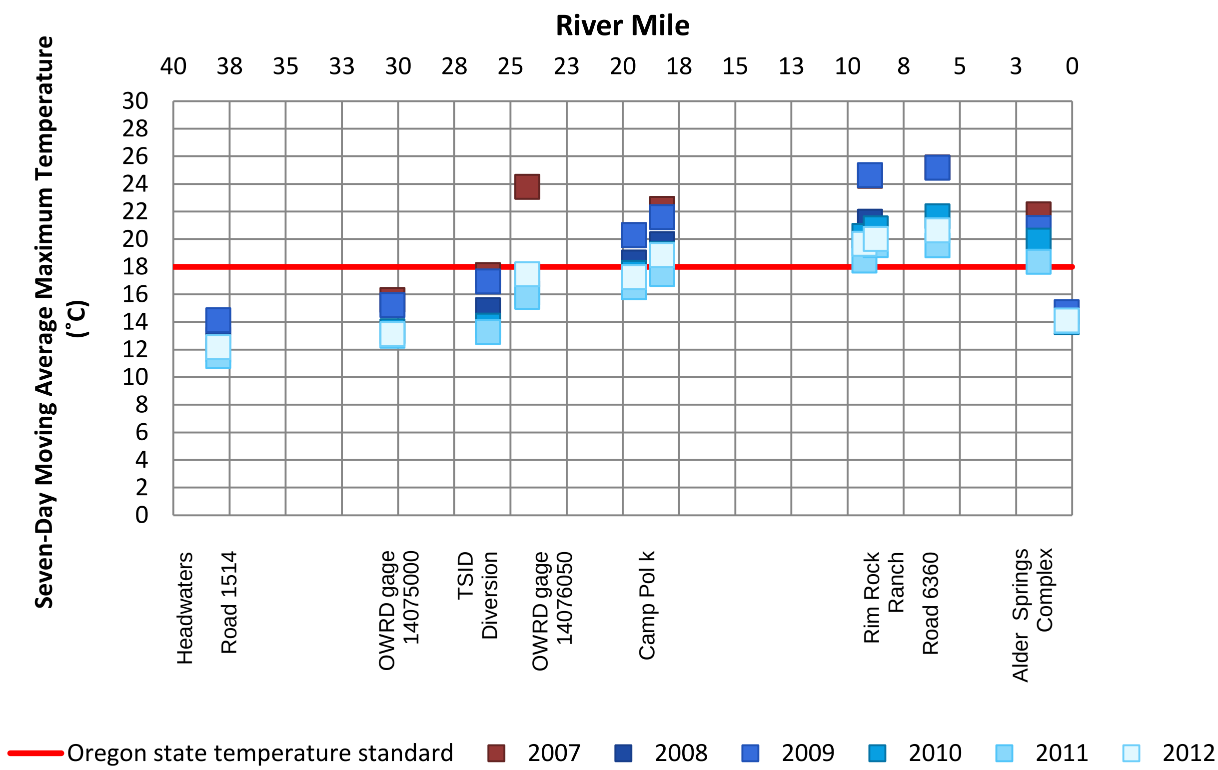


Figure 5.
The longitudinal temperature profile of Whychus Creek on the hottest water days from 2007 through 2012 shows warming from the headwaters to Alder Springs, where springs complex flows cool stream temperatures.

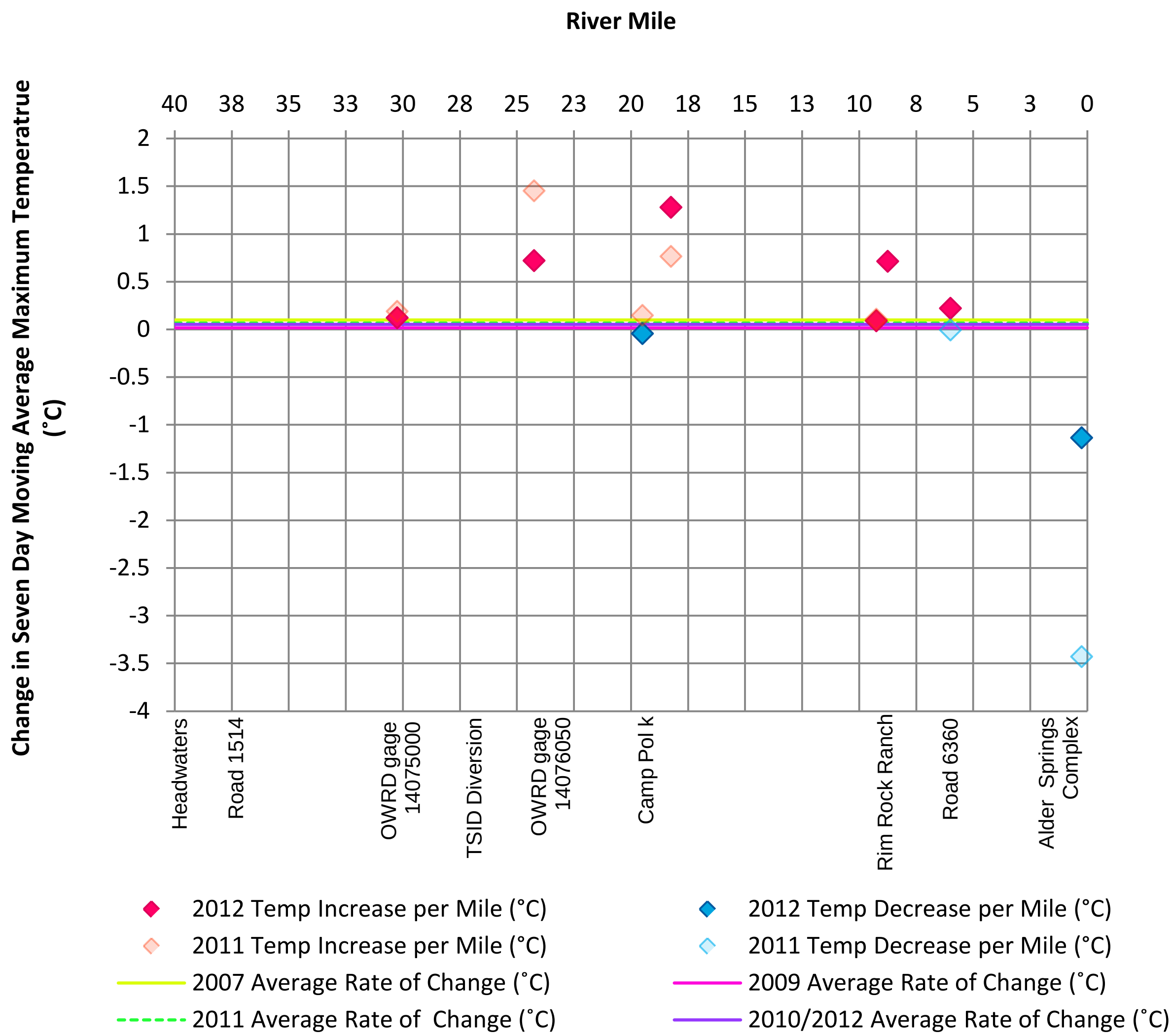


Figure 6.
Longitudinal average rate of temperature change in Whychus Creek in 2012 and 2011. Higher than average longitudinal changes in temperature identify reaches in which the rate of warming increased and suggest prioritization of these areas for research and restoration planning. Lower than average longitudinal changes in temperature highlight reaches where cooling occurred, indicating an opportunity to preserve optimal conditions through additional conservation measures

Target streamflow

Quadratic and cubic models were each statistically better than the lower-order model for WC 24.25 data (Table 3, Figure 8). Temperatures calculated from the WC 024.25 regression model suggest that 19 cfs (2.9 LnQD) was the minimum streamflow that produced a mean 7DMAX temperature at or below 18°C (± 1.7°C) given temperatures observed from July 2000-2012 at this location. The existing 33 cfs restoration target resulted in a mean 7DMAX temperature of 15.9°C ± 1.6°C (Appendix B), well below the 18°C standard. Although direct comparison to Heat Source model predictions is not possible because Heat Source uses the seven day moving average maximum temperatures, a daily statistic, and we use the mean seven day moving average maximum temperature for July, a monthly statistic, our 2000-2012 estimate for Sisters City Park differs only slightly from the 2008 Heat Source model which predicted a 7DMAX temperature of 15°C ± 1°C at 33 cfs at the ODFW gage at Sisters City Park (Watershed Sciences and MaxDepth Aquatics 2008). The 2000-2012 model estimates temperatures meeting the state standard at the same flow predicted by the 2000-2011 model and at a slightly lower flow than the 21 cfs predicted from earlier (2000-2008; 2000-2010) temperature-flow relationships.

For WC 006.00 data, the quadratic, cubic, and quartic regression models were each statistically better than the linear model. We evaluated temperatures calculated from both the cubic and quartic models at given flows for WC 006.00. At flows below 10 cfs/2.3 LnQD, temperature estimates from the two equations were within tenths of a degree, the exception being a 1.3°C difference in temperatures calculated for 2 cfs. Temperatures calculated from the quartic equation were consistently higher (more conservative) above 10 cfs than temperatures calculated from the cubic equation, with the difference in temperatures increasing through the range of flows from tenths of degrees at 10 cfs to more than two degrees at 154 cfs and topping out at a 5.3°C difference at the highest flows included in the analysis (210 cfs).The cubic model predicted 18°C at 53 cfs, lower than any previous model; the quartic model predicted 18°C at 75 cfs, more similar to previous models. Confidence intervals for the quartic model were also slightly lower than for the cubic model. We selected the quartic model as the more conservative model.

The quartic regression model for temperature-flow relationships at Road 6360 (WC 006.00) derived from 2000-2012 temperature and flow data predicts 75 cfs (4.0 LnQD) to be the minimum streamflow that will achieve a mean 7DMAX temperature of 18.0°C ± 1.8°C. Under this model the target streamflow of 33 cfs below Indian Ford Creek is projected to produce a mean 7DMAX temperature of 21.1°C ± 1.8°C at Road 6360, above the 18°C state standard but still below the 24°C lethal temperature threshold. (The cubic equation predicts 20.1°C± 1.9°C at the same flow.) The 2000-2012 quartic regression model predicts 18.6°C ± 1.8°C at 62 cfs; this result is remarkably similar to the Heat Source model estimate of 18.5°C ± 1°C at 62 cfs at Road 6360. This model also predicts temperatures meeting the state standard at flows similar to previous model estimates of 78 cfs (2000-2008) and 66 cfs (2000-2010, 2000-2011).

Table 3. A cubic regression provided the best fit to 2000-2012 temperature-flow data for WC 024.25.The quartic model provided the best fit to 2000-2012 data for WC 006.00. Temperatures calculated using the corresponding regression equations are expected to be the most accurate of the possible regression models.

Regression	Equation	df	S	R^2
	WC 024.25 (n=47)			
Linear	24.08-2.33(LnQD)	45	1.16	0.88
Quadratic	21.34-0.14(LnQD)-0.35$(LnQD)^2$	44	1.01	0.91
Cubic	**13.47+10.13(LnQD)-4.1$(LnQD)^2$ +0.4$(LnQD)^3$**	**43**	**0.61**	**0.97**
	WC 006.00 (n=46)			
Linear	28.44-2.53(LnQD)	44	1.52	0.86
Quadratic	22.82+2.02(LnQD)-0.75$(LnQD)^2$	43	1.0	0.93
Cubic	16.38+10.53(LnQD)-3.9$(LnQD)^2$ +0.34$(LnQD)^3$	42	0.76	0.96
Quartic	**23.63-2.77(LnQD)+4.05$(LnQD)^2$ -1.55$(LnQD)^3$+0.16$(LnQD)^4$**	**41**	**0.69**	**0.97**

a

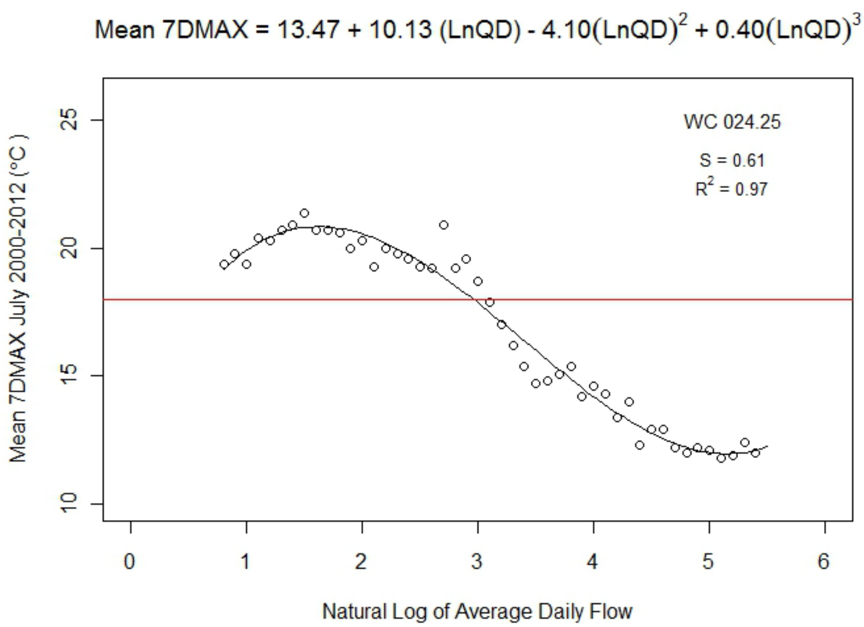
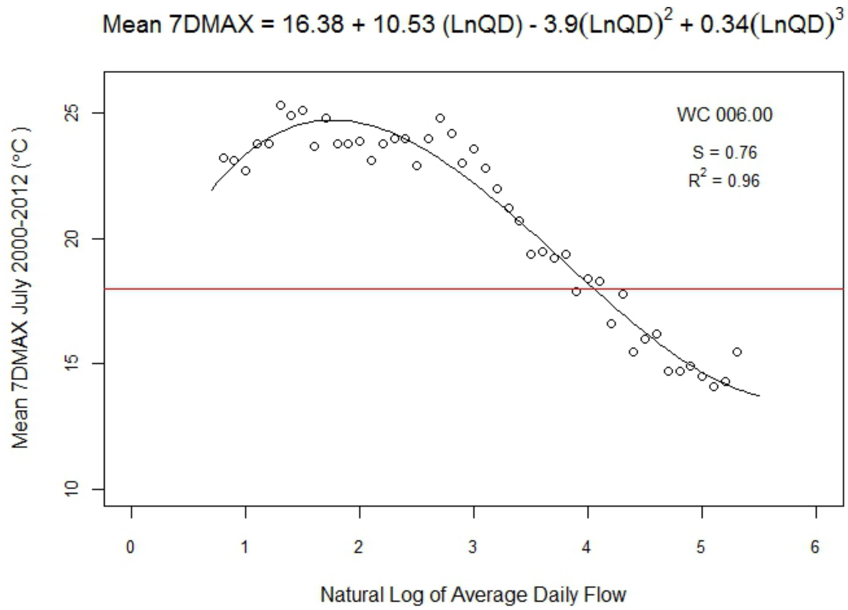
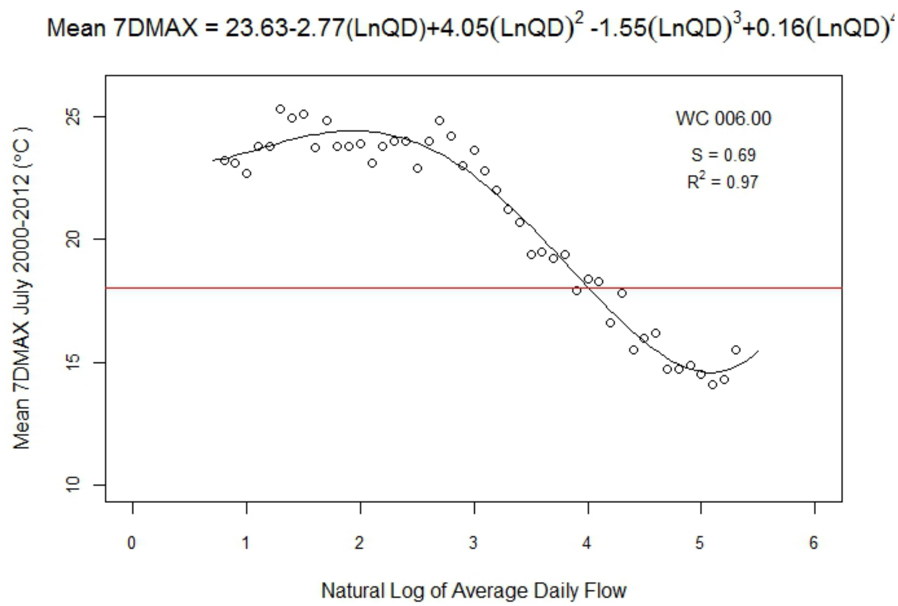


Figure 8.
Regression models fitted to temperature-flow data from July 2000-2012 describe the relationship between temperature and flow observed at a) Sister's City Park (WC 024.25), b) Road 6360 (WC 006.00), fitted with a cubic regression model, and c) Road 6360 fitted with a quartic regression model. Corresponding regression equations are used to calculate temperature at a given flow.

b


Mean 7DMAX = 16.38 + 10.53 (LnQD) - 3.9(LnQD)^2 + 0.34(LnQD)^3
WC 006.00
S = 0.76
R^2 = 0.96
Mean 7DMAX July 2000-2012 (°C)
Natural Log of Average Daily Flow
10
15
20
25
0
1
2
3
4
5
6

c


Mean 7DMAX = 23.63-2.77(LnQD)+4.05(LnQD)^2 -1.55(LnQD)^3+0.16(LnQD)^4
WC 006.00
S = 0.69
R^2 = 0.97
Mean 7DMAX July 2000-2012 (°C)
Natural Log of Average Daily Flow
10
15
20
25
0
1
2
3
4
5
6

Discussion

Temperature status

As flow conditions have improved, temperatures have exceeded the state standard for many fewer days and at fewer sites, with the highest temperatures occurring later in the summer. Although slightly higher than in 2011, temperatures in Whychus Creek in 2012 maintained the cooling trend observed over the last decade while continuing to exceed the state standard at some sites. While temperatures at Sisters City Park haven't exceeded the 18˚C state standard since 2008, temperatures at Road 6360 (WC 006.00) and at three additional sites (WC 008.75, WC 009.25, and WC 018.25) exceeded this criterion in 2012, consistent with the ODEQ 2010 303(d) Category 5 listing of Whychus Creek as water quality limited (ODEQ 2011). Whereas temperatures have historically exceeded the 18˚C state standard from as early as June into September and the hottest water days have occurred most frequently in July, in 2011 and 2012 temperatures exceeded the state standard only in August (with the exception of July 31st in 2012), and four of the five hottest water days over the past five years have occurred in August. Because the state standard is biologically based, we can infer that temperature conditions exceeding 18˚C were marginal to support salmon and trout rearing and migration. However, despite temperatures consistently exceeding optimal conditions at some sites, temperatures remained below the 24˚C lethal threshold for the third consecutive year.

Seven day moving average maximum temperatures for the dates for which data are available indicate suboptimal conditions for salmonid spawning. Although temperature conditions for salmon and trout spawning never exceeded the 12.8˚C biological requirement at Sister's City Park (WC 024.25) during the January 1 – May 15 spawning season in 2012, 7DMAX temperatures did exceed that threshold for about half of spawning season days for which data are available at all four downstream sites for which data are available (WC 006.00, WC 008.75, WC 009.25 and WC 018.25). Redband peak spawning season was not reported for 2012 due to highly ineffective redd counts resulting from high turbidity (Quesada et al. 2013); spawning peaked in May in three out of five years (2007, 2008 and 2010), but has also peaked in April (2009) and in June (2011). In 2011 the second highest proportion of redds was detected at Camp Polk, just upstream of WC 018.25 (the highest proportion was detected at Alder Springs, downstream of cold water inputs where temperature is unlikely to limit spawning). Temperatures at the four downstream sites continued to intermittently exceed 12.8˚C through June 2012. Temperature data were available for only four days and at only one site for April; while available temperature data did not exceed the spawning criterion before May 8, the first day for which temperature data are available, at any site, 7DMAX temperatures may have exceeded the 12.8˚C spawning requirement during redband spawning in April. Temperature dataloggers were deployed in early April 2013 to improve our ability to evaluate temperature conditions during redband spawning season.

Target streamflow

The state water right for Whychus Creek protects 20 cfs instream above Indian Ford Creek, just upstream of RM 20, and 33 cfs downstream of Indian Ford Creek. However, because no additional flows enter Whychus Creek between the headwaters and Indian Ford Creek, Deschutes River Conservancy has established a streamflow restoration target of 33 cfs for the entire length of the creek from headwaters to mouth. Although the 2000-2012 temperature-flow model predicts temperatures of 15.9°C ±1.6°C at 33 cfs at Sisters City Park, well below the state temperature standard and salmonid rearing and migration requirements, the estimate of 21.1°C ±1.8°C at 33 cfs at Road 6360 supports the conclusion of previous analyses that 33 cfs at Sisters City Park is insufficient to produce temperatures that comply with state standards for steelhead and salmon rearing and migration at this site. The temperature-flow relationship observed from 2000 to 2012 suggests that the 18°C state temperature standard will only be predictably met at Road 6360 at minimum flows of 75 cfs at Sisters City Park.

Despite summer temperatures exceeding 18°C along much of the lower reaches of Whychus Creek, reduced temperatures approaching 18°C in these reaches may nonetheless provide habitat conditions that will support steelhead and salmon rearing and migration, especially if other habitat features, such as adequate flow for steelhead outmigration and pools and cover for resident redband, are available. As early as 1895 irrigation diversions resulted in the dewatering of Whychus Creek near Sisters (Bob Main, personal communication c.f. Nehlsen 1995). As of 1950, a series of springs maintained flows of approximately 20 cfs below the dewatered reach from rm 15 to the mouth of the stream (Nielson 1950). As new water rights were issued, additional major sections of the creek became dewatered during the irrigation season (Mathisen 1985). These diminished flows supported up to 20 spawning chinook salmon or redds counted in the creek in 1952, and steelhead spawner numbers as high as 1000 in 1953 (Montgomery 1953). From these data we can infer that summer flows of 20 cfs in the reaches below rm 15 supported steelhead populations that produced up to 1000 spawning adults in 1953 and Chinook populations producing up to 20 spawning adults in 1952. Although the 18°C standard guarantees suitable temperatures for steelhead rearing and migration, the historical record of steelhead and salmon populations persisting in even lower flows than 33 cfs suggests the 21.1°C 7DMAX predicted at this level may be adequate to support steelhead and salmon rearing and migration.

Streamflow restoration effectiveness

Previous Whychus Creek water quality status reports have included BACI analyses to demonstrate cause and effect between cumulative increases in flow resulting from streamflow restoration and reduced temperatures in downstream reaches. Mork (2012a) presented results of a BACI analysis comparing differences in temperature in two restoration reaches relative to a reference reach in 2002 and in 2010. Median July streamflow increased by almost 30 cfs between 2002 and 2010, from 6.5 cfs to 35 cfs. If temperature has changed in response to streamflow, we expect to best be able to measure this change by comparing the more extreme difference in flows observed over this interval than is reflected in incremental flow increases from one year to the next. We expect the 28.5 cfs flow difference between 2002 and 2010 to be of sufficient magnitude to demonstrate any corresponding temperature response. Results from this analysis indicated a significant cooling response between WC 030.25, upstream of the TSID diversion, and WC 024.25 at Sisters City Park three miles downstream of the TSID diversion, and between the upstream site and WC 006.00 at Road 6360, relative to temperature changes in an upstream reference reach where no streamflow restoration occurred. Having demonstrated a cooling response to the increase in flow between 2002 and 2010, we feel confident that further increases in streamflow in Whychus will result in reductions in temperature. Accordingly, we do not present a BACI analysis for 2012 data.

Our 2012 BACI analysis of 2010-2011 data demonstrated the substantial influence of Alder Springs flows in maintaining low stream temperatures in downstream reaches, evidenced by temperatures increasing below Alder Springs when upstream flows increased (Mork 2012b). This result emphasizes the importance of Alder Springs flows in providing a relatively constant temperature refuge, and highlights the potential impacts of groundwater withdrawals on stream conditions in this area as well as in the Deschutes below the confluence with Whychus. Despite limited knowledge of the specific dynamics of the hydrology of Alder Springs, the most comprehensive synthesis of available groundwater and surface water data states that "virtually all groundwater not consumptively used in the Upper Deschutes Basin discharges to the stream system upstream of the vicinity of Pelton Dam. . . Groundwater and surface water are, therefore, directly linked, and removal of groundwater will ultimately diminish streamflow"

(USGS 2001). An improved scientific understanding of the hydrology of Alder Springs and the anticipated effects of groundwater withdrawals will allow conservation and restoration partners to better plan to address these effects in the future.

Conclusions

Although stream temperatures in Whychus continue to exceed the state standard, and despite a slight uptick in stream temperature in 2012, the number of days on which temperatures in Whychus Creek exceeded the state standard was down by half in 2012 from 2009 and 2010 numbers. Temperatures in Whychus exceeded the state standard for only six days more in 2012 than in 2011, resulting in almost an additional month in each of the past two years during which temperature conditions were suitable for fish along the entire length of Whychus.

The temperature-flow relationship described by twelve years of data suggests that 33 cfs at Sisters City Park (WC 024.25) is more than sufficient to meet the 18°C salmon and trout rearing and migration requirement. Eighteen miles downstream at Road 6360 (WC 006.00), 75 cfs is the minimum flow estimated to meet the 18°C standard. Although 75 cfs may not currently be a feasible restoration target, these data nonetheless provide a benchmark for streamflow restoration. Other restoration actions, such as stream channel restoration projects at Camp Polk Meadow Preserve (RM 18 – 19.5) and Whychus Canyon (RM 9 - 15), will further promote cooling in the temperature-impaired reach. Given suitable temperatures in cooler upstream reaches, rearing salmon and trout may survive to migrate through warmer waters to the Deschutes. Our results show that streamflow restoration has already improved temperature conditions for re-introduced salmon and trout, and advance our understanding of temperature and flow on Whychus Creek to better inform future watershed restoration efforts.

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APPENDIX A Whychus Creek continuous temperature monitoring stations 1995-2011. TE = Temperature status, RE = Restoration effectiveness, ST = Streamflow target.

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

APPENDIX B Temperatures at given flows.

Whychus Creek at Sister's City Park (WC 024.25) at flows from 2.4 cfs to 222 cfs.

Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)
2	18.7	1.7	**57**	13.8	1.6	**112**	12.0	1.5	**167**	11.5	1.5
3	20.2	1.7	**58**	13.8	1.6	**113**	12.0	1.5	**168**	11.5	1.5
4	20.7	1.7	**59**	13.7	1.6	**114**	12.0	1.5	**169**	11.5	1.5
5	20.8	1.7	**60**	13.7	1.6	**115**	12.0	1.5	**170**	11.5	1.5
6	20.8	1.7	**61**	13.6	1.6	**116**	11.9	1.5	**171**	11.5	1.5
7	20.6	1.7	**62**	13.6	1.6	**117**	11.9	1.5	**172**	11.5	1.5
8	20.4	1.7	**63**	13.5	1.6	**118**	11.9	1.5	**173**	11.5	1.5
9	20.2	1.7	**64**	13.5	1.6	**119**	11.9	1.5	**174**	11.5	1.5
10	19.9	1.7	**65**	13.4	1.6	**120**	11.9	1.5	**175**	11.5	1.5
11	19.7	1.7	**66**	13.4	1.6	**121**	11.9	1.5	**176**	11.5	1.5
12	19.5	1.7	**67**	13.3	1.6	**122**	11.9	1.5	**177**	11.5	1.5
13	19.2	1.7	**68**	13.3	1.6	**123**	11.8	1.5	**178**	11.5	1.5
14	19.0	1.7	**69**	13.2	1.6	**124**	11.8	1.5	**179**	11.5	1.5
15	18.8	1.7	**70**	13.2	1.6	**125**	11.8	1.5	**180**	11.5	1.5
16	18.6	1.7	**71**	13.1	1.6	**126**	11.8	1.5	**181**	11.5	1.5
17	18.4	1.7	**72**	13.1	1.6	**127**	11.8	1.5	**182**	11.5	1.5
18	18.2	1.7	**73**	13.1	1.6	**128**	11.8	1.5	**183**	11.5	1.5
19	18.0	1.7	**74**	13.0	1.6	**129**	11.8	1.5	**184**	11.5	1.5
20	17.8	1.7	**75**	13.0	1.6	**130**	11.8	1.5	**185**	11.5	1.5
21	17.6	1.6	**76**	12.9	1.6	**131**	11.8	1.5	**186**	11.5	1.5
22	17.4	1.6	**77**	12.9	1.6	**132**	11.7	1.5	**187**	11.5	1.5
23	17.3	1.6	**78**	12.9	1.6	**133**	11.7	1.5	**188**	11.5	1.5
24	17.1	1.6	**79**	12.8	1.6	**134**	11.7	1.5	**189**	11.5	1.5
25	16.9	1.6	**80**	12.8	1.6	**135**	11.7	1.5	**190**	11.5	1.5
26	16.8	1.6	**81**	12.8	1.6	**136**	11.7	1.5	**191**	11.5	1.5
27	16.6	1.6	**82**	12.7	1.6	**137**	11.7	1.5	**192**	11.5	1.5
28	16.5	1.6	**83**	12.7	1.6	**138**	11.7	1.5	**193**	11.5	1.5
29	16.4	1.6	**84**	12.7	1.6	**139**	11.7	1.5	**194**	11.5	1.5
30	16.2	1.6	**85**	12.6	1.6	**140**	11.7	1.5	**195**	11.5	1.5
31	16.1	1.6	**86**	12.6	1.6	**141**	11.7	1.5	**196**	11.5	1.5
32	16.0	1.6	**87**	12.6	1.6	**142**	11.7	1.5	**197**	11.5	1.5
33	15.9	1.6	**88**	12.5	1.6	**143**	11.7	1.5	**198**	11.5	1.5
34	15.7	1.6	**89**	12.5	1.6	**144**	11.6	1.5	**199**	11.5	1.5
35	15.6	1.6	**90**	12.5	1.6	**145**	11.6	1.5	**200**	11.5	1.5
36	15.5	1.6	**91**	12.5	1.6	**146**	11.6	1.5	**201**	11.5	1.5
37	15.4	1.6	**92**	12.4	1.6	**147**	11.6	1.5	**202**	11.5	1.5
38	15.3	1.6	**93**	12.4	1.6	**148**	11.6	1.5	**203**	11.5	1.5
39	15.2	1.6	**94**	12.4	1.6	**149**	11.6	1.5	**204**	11.5	1.5
40	15.1	1.6	**95**	12.4	1.6	**150**	11.6	1.5	**205**	11.6	1.5
41	15.0	1.6	**96**	12.3	1.5	**151**	11.6	1.5	**206**	11.6	1.5
42	14.9	1.6	**97**	12.3	1.5	**152**	11.6	1.5	**207**	11.6	1.5
43	14.9	1.6	**98**	12.3	1.5	**153**	11.6	1.5	**208**	11.6	1.5
44	14.8	1.6	**99**	12.3	1.5	**154**	11.6	1.5	**209**	11.6	1.5
45	14.7	1.6	**100**	12.2	1.5	**155**	11.6	1.5	**210**	11.6	1.5
46	14.6	1.6	**101**	12.2	1.5	**156**	11.6	1.5	**211**	11.6	1.5
47	14.5	1.6	**102**	12.2	1.5	**157**	11.6	1.5	**212**	11.6	1.5
48	14.4	1.6	**103**	12.2	1.5	**158**	11.6	1.5	**213**	11.6	1.5
49	14.4	1.6	**104**	12.2	1.5	**159**	11.6	1.5	**214**	11.6	1.5
50	14.3	1.6	**105**	12.1	1.5	**160**	11.6	1.5	**215**	11.6	1.5
51	14.2	1.6	**106**	12.1	1.5	**161**	11.6	1.5	**216**	11.6	1.5
52	14.2	1.6	**107**	12.1	1.5	**162**	11.6	1.5	**217**	11.6	1.5
53	14.1	1.6	**108**	12.1	1.5	**163**	11.6	1.5	**218**	11.6	1.5
54	14.0	1.6	**109**	12.1	1.5	**164**	11.6	1.5	**219**	11.6	1.5
55	14.0	1.6	**110**	12.0	1.5	**165**	11.5	1.5	**220**	11.6	1.5
56	13.9	1.6	**111**	12.0	1.5	**166**	11.5	1.5	**221**	11.6	1.5

Whychus Creek at Road 6360 (WC 006.00) at flows from 2.4 cfs to 209 cfs.

Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)	Flow (cfs)	Mean Temp (7DMAX)	CI (±)
2	23.2	1.9	**57**	18.9	1.8	**112**	17.2	1.8	**167**	17.5	1.8
3	23.7	1.9	**58**	18.9	1.8	**113**	17.2	1.8	**168**	17.5	1.8
4	24.0	1.9	**59**	18.8	1.8	**114**	17.2	1.8	**169**	17.6	1.8
5	24.3	1.9	**60**	18.8	1.8	**115**	17.2	1.8	**170**	17.6	1.8
6	24.4	1.9	**61**	18.7	1.8	**116**	17.2	1.8	**171**	17.6	1.8
7	24.4	1.9	**62**	18.6	1.8	**117**	17.2	1.8	**172**	17.6	1.8
8	24.4	1.9	**63**	18.6	1.8	**118**	17.2	1.8	**173**	17.6	1.8
9	24.4	1.9	**64**	18.5	1.8	**119**	17.2	1.8	**174**	17.6	1.8
10	24.3	1.9	**65**	18.5	1.8	**120**	17.2	1.8	**175**	17.7	1.8
11	24.2	1.9	**66**	18.4	1.8	**121**	17.2	1.8	**176**	17.7	1.8
12	24.1	1.9	**67**	18.4	1.8	**122**	17.2	1.8	**177**	17.7	1.8
13	23.9	1.9	**68**	18.3	1.8	**123**	17.2	1.8	**178**	17.7	1.8
14	23.8	1.9	**69**	18.3	1.8	**124**	17.2	1.8	**179**	17.7	1.8
15	23.7	1.9	**70**	18.2	1.8	**125**	17.2	1.8	**180**	17.8	1.8
16	23.5	1.9	**71**	18.2	1.8	**126**	17.2	1.8	**181**	17.8	1.8
17	23.4	1.9	**72**	18.1	1.8	**127**	17.2	1.8	**182**	17.8	1.8
18	23.2	1.9	**73**	18.1	1.8	**128**	17.2	1.8	**183**	17.8	1.8
19	23.0	1.9	**74**	18.1	1.8	**129**	17.2	1.8	**184**	17.8	1.8
20	22.9	1.9	**75**	18.0	1.8	**130**	17.2	1.8	**185**	17.9	1.8
21	22.7	1.9	**76**	18.0	1.8	**131**	17.2	1.8	**186**	17.9	1.8
22	22.6	1.9	**77**	17.9	1.8	**132**	17.2	1.8	**187**	17.9	1.8
23	22.4	1.9	**78**	17.9	1.8	**133**	17.2	1.8	**188**	17.9	1.8
24	22.3	1.9	**79**	17.9	1.8	**134**	17.2	1.8	**189**	17.9	1.8
25	22.2	1.9	**80**	17.8	1.8	**135**	17.2	1.8	**190**	18.0	1.8
26	22.0	1.9	**81**	17.8	1.8	**136**	17.2	1.8	**191**	18.0	1.8
27	21.9	1.9	**82**	17.8	1.8	**137**	17.2	1.8	**192**	18.0	1.8
28	21.7	1.8	**83**	17.7	1.8	**138**	17.2	1.8	**193**	18.0	1.8
29	21.6	1.8	**84**	17.7	1.8	**139**	17.2	1.8	**194**	18.1	1.8
30	21.5	1.8	**85**	17.7	1.8	**140**	17.2	1.8	**195**	18.1	1.8
31	21.4	1.8	**86**	17.6	1.8	**141**	17.2	1.8	**196**	18.1	1.8
32	21.2	1.8	**87**	17.6	1.8	**142**	17.2	1.8	**197**	18.1	1.8
33	21.1	1.8	**88**	17.6	1.8	**143**	17.2	1.8	**198**	18.1	1.8
34	21.0	1.8	**89**	17.6	1.8	**144**	17.2	1.8	**199**	18.2	1.8
35	20.9	1.8	**90**	17.5	1.8	**145**	17.2	1.8	**200**	18.2	1.8
36	20.8	1.8	**91**	17.5	1.8	**146**	17.3	1.8	**201**	18.2	1.8
37	20.7	1.8	**92**	17.5	1.8	**147**	17.3	1.8	**202**	18.2	1.8
38	20.6	1.8	**93**	17.5	1.8	**148**	17.3	1.8	**203**	18.3	1.8
39	20.4	1.8	**94**	17.5	1.8	**149**	17.3	1.8	**204**	18.3	1.8
40	20.3	1.8	**95**	17.4	1.8	**150**	17.3	1.8	**205**	18.3	1.8
41	20.2	1.8	**96**	17.4	1.8	**151**	17.3	1.8	**206**	18.3	1.8
42	20.1	1.8	**97**	17.4	1.8	**152**	17.3	1.8	**207**	18.4	1.8
43	20.1	1.8	**98**	17.4	1.8	**153**	17.3	1.8	**208**	18.4	1.8
44	20.0	1.8	**99**	17.4	1.8	**154**	17.3	1.8	**209**	18.4	1.8
45	19.9	1.8	**100**	17.3	1.8	**155**	17.4	1.8	**210**	18.4	1.8
46	19.8	1.8	**101**	17.3	1.8	**156**	17.4	1.8			
47	19.7	1.8	**102**	17.3	1.8	**157**	17.4	1.8			
48	19.6	1.8	**103**	17.3	1.8	**158**	17.4	1.8			
49	19.5	1.8	**104**	17.3	1.8	**159**	17.4	1.8			
50	19.5	1.8	**105**	17.3	1.8	**160**	17.4	1.8			
51	19.4	1.8	**106**	17.3	1.8	**161**	17.4	1.8			
52	19.3	1.8	**107**	17.3	1.8	**162**	17.4	1.8			
53	19.2	1.8	**108**	17.2	1.8	**163**	17.5	1.8			
54	19.2	1.8	**109**	17.2	1.8	**164**	17.5	1.8			
55	19.1	1.8	**110**	17.2	1.8	**165**	17.5	1.8			
56	19.0	1.8	**111**	17.2	1.8	**166**	17.5	1.8			

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 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 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 2010 one barrier was retrofitted to provide fish passage, connecting two reaches and reducing total reaches to six. In 2011 UDWC retrofitted a second barrier to restore passage, and determined that a third barrier identified in the baseline inventory does not restrict fish passage. At the end of 2011 the total number of fragmented reaches below natural barriers had been reduced to four, varying in length from two to 20 miles. No new fish passage projects were completed in 2012. Removal of of a fourth barrier originally slated for September 2012 will instead be implemented in fall 2013 due to a delay in federal funding. Conversations are ongoing with water rights holders to address fish passage at a fifth barrier. UDWC will continue to actively engage water rights holders to provide passage at the final two barriers by 2020. Removal of the three remaining barriers will provide access to an additional 16 river miles and restore connectivity within the full length of stream habitat historically accessible to resident and anadromous species.

Introduction

The extent of stream connectivity, as influenced by the existence, condition and location of fish passage barriers, was selected as an indicator to be tracked over time on Whychus Creek. Although stream connectivity can be influenced by poor water quality or other habitat conditions fish passage barriers are the primary feature affecting connectivity in Whychus Creek. This technical report presents the status of connectivity and progress made on restoring fish passage at the close of 2012.

Fish passage barriers are widely recognized as hindering habitat connectivity by obstructing movement of aquatic species with the presence of physical barriers, changing velocities, water quality conditions and overall hydraulic and thermal alterations (Bergkamp *et al* 2000). With this recognition comes the realization that habitat connectivity along river systems is essential to healthy ecological function (Cote *et al* 2009, Wiens 2002).

Passage barriers are therefore a simple and effective indicator of determining how much habitat is available to resident and anadromous fish species in Whychus Creek (Cote *et al* 2009). UDWC and its partners are working with landowners and water right holders to provide unimpeded up- and down-stream fish passage by retrofitting or removing all fish passage barriers in Whychus Creek by 2020.

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

UDWC used OWRD data and collected new data to establish a baseline number of barriers in Whychus Creek not meeting state and federal fish passage criteria. We calculated the number of fragmented reaches created by the barriers and the number of stream miles for each reach. This report presents the status of fish passage and stream connectivity at the close of 2012.

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 all four 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 reduce 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) appears not to pose a barrier to fish passage. 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, are accessible to fish, and a total of four fragmented reaches remain below natural barriers.

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

Over the course of 2012 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 2012, conversations were ongoing with water rights holders to address fish passage at a fifth barrier, No. 4, at rm 22.2.

Table 1.
Passage barrier specifications and status as of 2012. 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]	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.	
No. 3	4/3/2009	22.6	44.282	-121.5531	100%	2.5	36.0	18.0	Yes	Sokol dam once used to create a backwater for fish rearing. No longer used and not associated with an irrigation water right.	Oct-11
No. 4	4/3/2009	23.8	44.2678	-121.5584	100%	4.5	48.0	18.0	Yes	Sokol irrigation diversion dam.	
No. 5	8/28/2009	25.2	44.2515	-121.5502	100%	N/A	≤ 6.0	N/A	Yes	Three Sisters Irrigation District Dam. Channel raised to dam height and riffle created. Tallest feature height ≤ 6.0.	Dec-10
No. 6	8/28/2009	26.8	44.2356	-121.5633	100%	3.2	45.0	43.0	Yes	McCallister irrigation diversion dam	

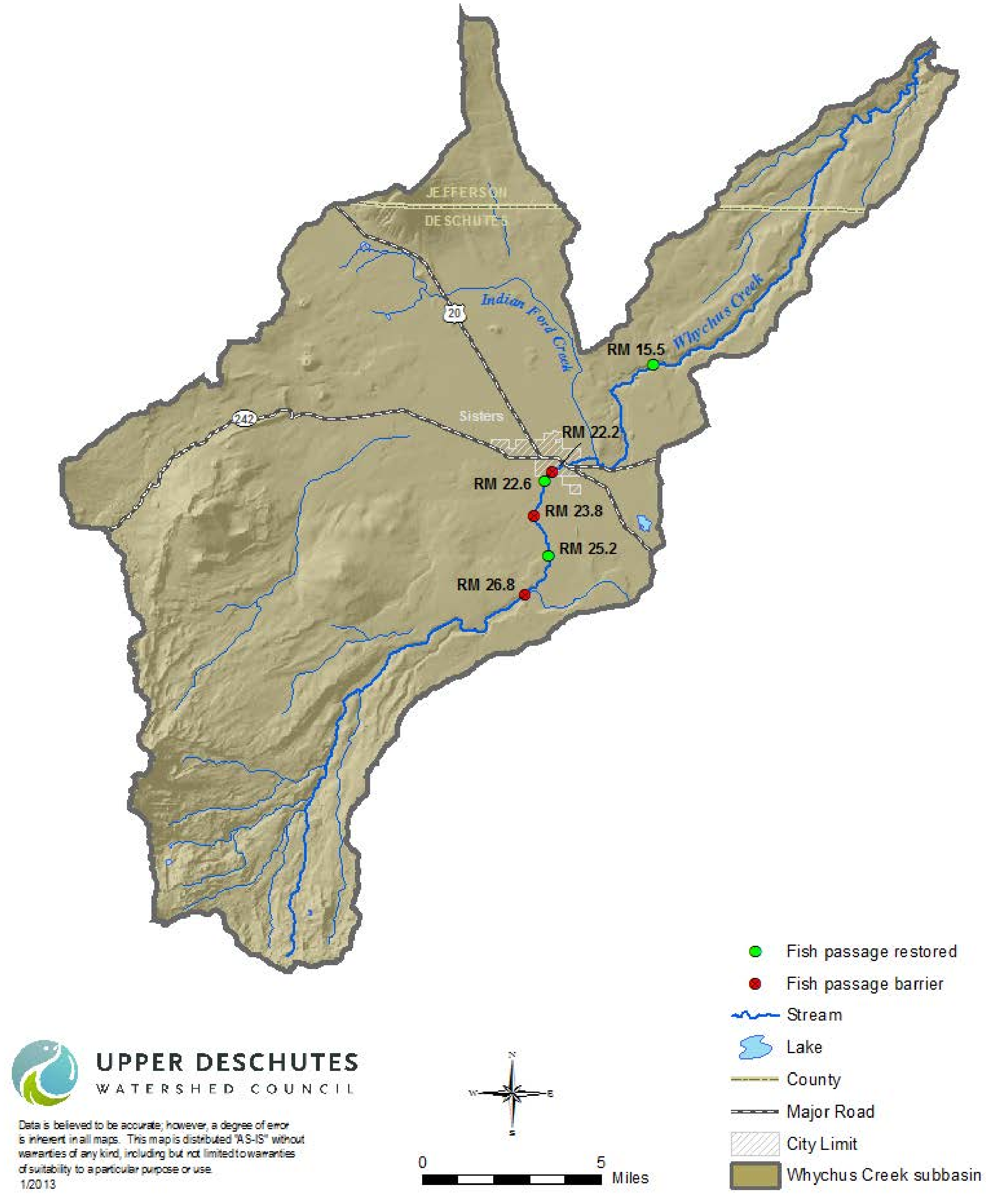

Figure 1.
In 2009, six barriers impaired stream connectivity on Whychus Creek between river miles 15.5 and 26.8. From 2009 to 2012 fish passage was restored or found to be intact at three of these barriers. UDWC and restoration partners aim to provide passage at each of these barriers by 2020.

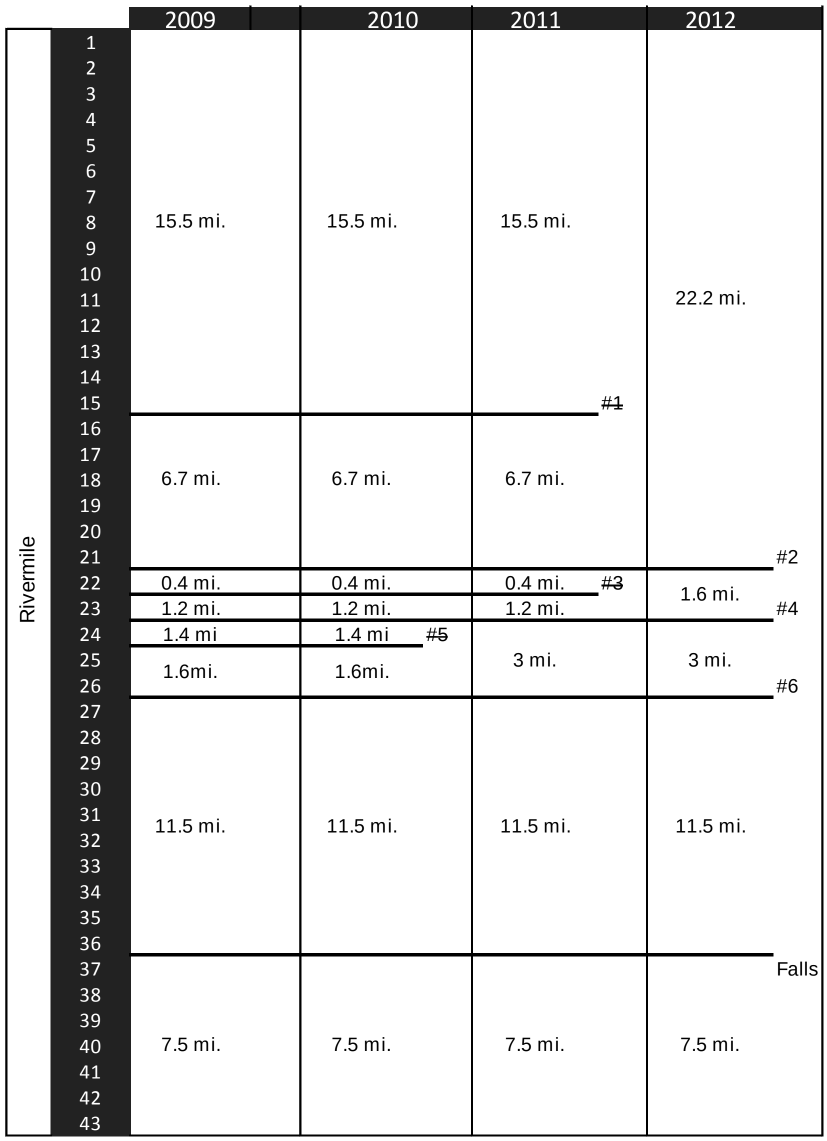

Figure 2.

Three barriers to fish passage remain as of 2012, reducing the number of fragmented reaches from a baseline of seven to four, varying in length from 1.6 to 22.2 miles. Whychus Creek Falls, located between river miles 36 and 37, is a natural barrier to fish passage.

Discussion

Existing barriers determine the number of miles of contiguous stream habitat accessible to fish. Habitat connectivity has increased as barriers have been removed. UDWC will continue to partner with water rights holders to develop and implement projects that will restore fish passage at the remaining three barriers by 2020. Removal of these barriers will provide access to an additional 16 river miles and restore connectivity along the entire length of stream habitat historically accessible to resident and anadromous species. Tracking miles of habitat newly opened to fish will continue to provide important context for the recovery of anadromous and resident fish populations in Whychus Creek.

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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. From 2009-2010 restoration partners screened two diversions, reducing cumulative unscreened diverted flows from 193 to 190.43 cfs. This number was further reduced to 175.45 through water transactions conducted by Deschutes River Conservancy (DRC). In 2011, through cooperation with the Three Sisters Irrigation District (TSID), the TSID diversion was screened, achieving an 80% reduction from 2009 baseline unscreened flows. As of 2011, only 37.77 of the total 178.02 cfs diverted for irrigation remain unscreened. This number was reduced by an additional 5.52 cfs in 2012 through consolidation of an upstream point of diversion with TSID to a new low of 32.25. Although potential for fish entrainment in irrigation diversions has thus been substantially reduced, UDWC and restoration partners remain committed to continuing to engage with water rights holders and landowners to eliminate all risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

Introduction

UDWC selected fish entrainment potential as an indicator of restoration effectiveness on Whychus Creek. UDWC described fish entrainment potential by the presence and location of irrigation diversions lacking state and federally approved fish screens. Irrigation diversions can create two types of problems for fish. First, they potentially block upstream and downstream fish passage. Second, unscreened diversions divert fish almost as effectively as they divert water. This technical report updates the potential for fish entrainment at irrigation diversions in Whychus Creek at the close of 2012.

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). As of 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, 98.5% of the 2010 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 represented 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 that although the diversion, a small-diameter pipe, does not meet screening criteria, the insignificant volume diverted (0.05 cfs) and the flow-through course by which diverted flows are returned to the creek render the threat to fish survival posed by this diversion minimal (R. Houston, personal communication 2012).

Water rights associated with the Lazy Z/Uncle John diversion were found to be 9.3 cfs, higher than the 5.52 accounted for during the 2009 baseline inventory. 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 to 0.00 and transferred 1.61 cfs instream; ongoing water rights transactions will transfer an additional 7.69 cfs of Leithauser water rights to TSID. The Lazy Z/Uncle John diversion (Diversion No. 3, rm 25.5) was decommissioned and the point of diversion consolidated with TSID, eliminating the risk of fish entrainment at this site. This consolidation reduced the number of diversions not meeting screening criteria to eight, and reduced unscreened flows by 9.3 cfs. Between water rights transactions and screening of irrigation diversions, the cumulative unscreened diversion rate was reduced to a new low of 32.25 in 2012. Water rights transactions further reduced diversion rates by an additional 1.00 cfs at Leithauser, however, because Leithauser had already been screened, this reduction did not affect the cumulative unscreened diversion rate.

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 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 2012 conversations are ongoing to screen a sixth diversion on Whychus Creek.

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 from 2009-2012, leaving eight diversions unscreened.

2002-2009 Baseline data											
Diversion ID	Baseline Sampling Date	River Mile	Diversion Type	2009 Associated Diversion Rate (cfs)	Screen Present at Baseline Inventory	Screen opening size (inches)	Met State & Federal Criteria at Baseline Inventory	2012 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		No	Plainview. Junior water rights. Diversion rarely on
No. 2	8/28/2009	25.15	Gravity	21.59	No	N/A	No	21.59		No	McCallister
No. 3	8/28/2009	23.90	Gravity	5.52	No	N/A	No	0.00	May-12	Yes	Lazy Z / Uncle John
No. 4	8/28/2009	23.65	Gravity	153.00	No	N/A	No	133.68	Apr-11	Yes	TSID
No. 5	8/28/2009	23.65	Gravity	1.00	No	N/A	No	1.00	Oct-10	Yes	Edgington
No. 6	8/28/2009	22.30	Gravity	5.00	No	N/A	No	5.00		No	Sokol
No. 7	8/28/2009	20.90	Gravity	1.12	No	N/A	No	0.00	Oct-09	Yes	Leithauser
No. 8	8/28/2009	18.65	Pump	0.07	Yes	1/4	No	0.07		No	No. 9 on OWRD List
No. 9	8/28/2009	18.15	Pump	0.38	Yes	1/4	No	0.38		No	Bradley
No. 10	8/28/2009	17.50	Pump	0.45	Yes	3/32	Yes	0.45	Aug-09	Yes	Deggendorfer
No. 11	9/30/2002	14.75	Pump	0.05	Yes	No Data	No	0.05		No	Meyer. Fish screening assessed by OWRD
No. 12	9/24/2002	11.20	Gravity	0.68	No	N/A	No	0.68		No	Remund
No. 13	9/24/2002	9.25	Gravity	0.60	No	N/A	No	0.60		No	Baker
			Baseline Diversion Total	**193.34**			**2012 Diversion Total**	**167.38**			
			Baseline Unscreened Total	**192.89**			**2012 Unscreened Total**	**32.25**			

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

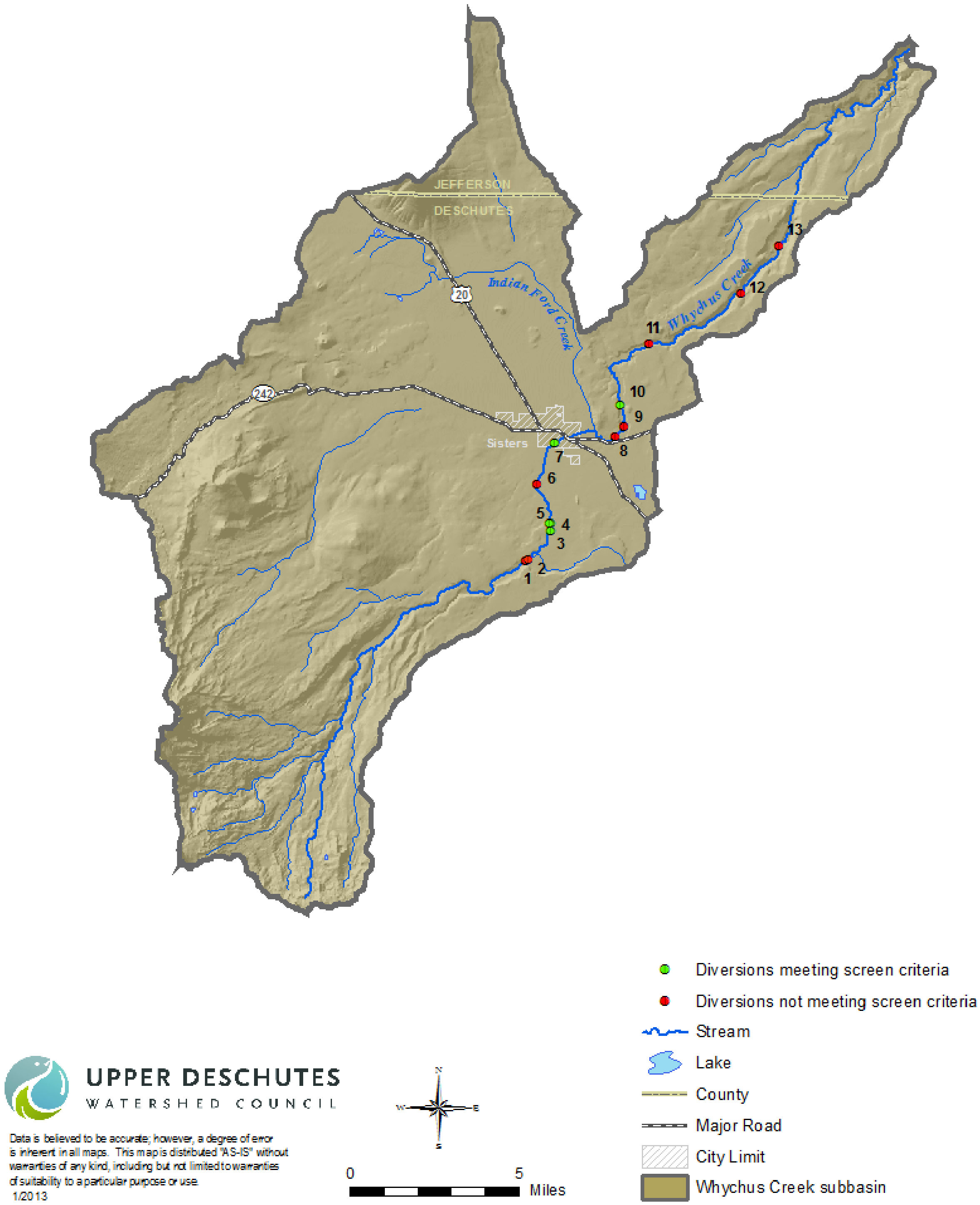

Figure 1.
In 2009 UDWC identified 13 irrigation diversions on Whychus Creek of which 12 did not meet federal and state criteria for fish screens (NMFS 2008). One diversion identified during baseline data collection met ODFW and NMFS screening criteria. From 2009 to 2012 four diversions were screened or decommissioned to reduce cumulative unscreened flows to 32.25. Diversions No. 4 and 5 are located within feet of each other on opposite sides of the creek, thus only one of the two points is visible on the map.

Discussion

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

As of the close of 2012, the cumulative unscreened diversion rate had been reduced by 83%, from 192.89 in 2009 to 32.25 cfs in 2012. Potential for fish entrainment in irrigation diversions has thus been substantially reduced, owed in large part to the progressive practices of TSID management. UDWC and restoration partners remain committed to continuing to engage with water rights holders and landowners to eliminate all risk of entrainment by meeting screening criteria at all diversions on Whychus Creek.

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Effectiveness Monitoring in Whychus Creek; Benthic Macroinvertebrate Communities in 2005, 2009, 2011, and 2012

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Abstract

Aquatic macroinvertebrate monitoring was done in 2005, 2009, 2011, and 2012 at 10-13 sites along Whychus Creek (from RM 30.25 to RM 0.5), to determine baseline biological conditions and assess the effects of restoration on stream biota. Xerces staff trained volunteers in standardized DEQ wadeable stream sampling techniques, after which teams dispersed to collect samples at pre-determined sites. The high level of similarity among replicate site samples taken for quality assurance in each year indicates that trained volunteers implemented the protocol successfully, such that differences between sites and years are unlikely to be the result of operator error. In the past four years, the benthic macroinvertebrate community in Whychus Creek underwent substantial changes from 2005 to 2009, and showed increasing stabilization from 2009 to 2012. Biotic conditions indicated by PREDATOR and IBI scores show improvement followed by a period of little change among sites in downstream reaches of the creek, and greater fluctuation among sampling sites in the mid-reach portions of the stream. PREDATOR scores suggest a downward trend in biotic condition among upstream sampling sites, although the differences between years are not significant, but this may be an early warning of new or increased stressors. In contrast, IBI scores among upstream reach sampling sites show steady and significant improvement; these changes may be driven by the strong increase over time in richness, relative diversity, and abundance of sensitive EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) at upstream sites. PREDATOR and IBI scores at sites within the mid-reach region showed the greatest fluctuation over time, and are likely to continue to change as the stream responds to the recent relocation to its restored channel at Camp Polk. Both mean temperature and sediment optima of the macroinvertebrate assemblages sampled are consistently highest at downstream sites and lowest at upstream sites. Examination of the missing and replacement taxa identified by PREDATOR shows that in every year, mean fine sediment optima of replacement taxa is significantly lower than that of the missing taxa, while differences in mean temperature optima for missing vs. replacement taxa were significant in only a single year. Multiple years of effectiveness monitoring along Whychus Creek have enabled detection of longer term changes and trends among the benthic macroinvertebrate community, and data suggest that the invertebrate communities in many parts of the stream may be stabilizing. It would be of interest to conduct additional monitoring to determine whether the community composition among the mid-reach sampling sites continues to fluctuate, and if the trend towards decreased PREDATOR scores among upstream sites persists.

Project Background

Biomonitoring in Whychus Creek

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

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

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

Whychus Creek experienced significant habitat degradation in past decades from surrounding land use practices, including dewatering for irrigation, channelization, grazing, and stream-side development. This project was conducted as 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). The Xerces Society worked with Upper Deschutes Watershed Council to collect benthic macroinvertebrate samples along Whychus Creek in 2005, 2009, 2011, and 2012 from sites spanning RM 30.25 to RM 0.5. Sampling in 2005 was done prior to large-scale habitat restoration and before some stream flow restoration. This provided baseline data on existing macroinvertebrate communities along the stream, which in previous years would frequently go dry in summer. Sampling was repeated in 2009 and 2011 to assess the macroinvertebrate community after large scale stream flow restoration had been done, including the restoration of about 20 cfs of water to the creek flow, but prior to the planned restoration of channelized portions of the creek at Camp Polk Meadows. Sampling in 2012 was done after the restoration at Camp Polk meadows was implemented, although monitoring and maintenance of this restoration site is still ongoing.

Biotic assessment techniques

Predictive models

Assessment of biological communities is frequently done using two major analytical approaches: predictive models and multimetric Indices of Biological Integrity (IBI). Predictive models compare the macroinvertebrate community at a given sampling site to the community present at reference or best available-condition streams in the same region with similar physical, chemical, and biological characteristics (Wright *et al*. 2000). The predictive model PREDATOR (Predictive Assessment Tool for Oregon; Hubler 2008) was developed for two major regions in Oregon: the Marine Western Coastal Forest predictive model (Willamette Valley and Coast Range ecoregions) and the Western Cordillera and Columbia Plateau predictive model (Klamath Mountain, Cascades, East Cascades, Blue Mountains, and Columbia Plateau ecoregions). The model calculates the ratio of the taxa observed at a sampling site to the taxa expected at that site (O over E) based on data collected previously from a large number of reference streams. In general, an O/E value of less than one indicates loss of common taxa, while values greater than one may indicate taxa enrichment, potentially in response to pollution or nutrient loading. The model output also generates O/E scores for individual taxa at each sampling site, allowing specific taxa loss and replacement to be investigated.

While often considered to be more sensitive and accurate than multimetric assessments (see below), it should be noted that the PREDATOR model has not been recalibated since its inception, using data from streams surveyed in 1998-2004. Periodic sampling of reference streams used to build the models and model recalibration 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 altered over time, especially as the WC+CP (Western Cordillera+Columbia Plateau) predictive model applicable to the Cascades is known to have lower precision than the MWCF (Marine Western Coastal Forest) model developed for the Willamette Valley and Coast Range (Hubler 2008). In addition, while a PREDATOR score indicating "most disturbed" can be reliably assumed to reflect biologically disturbed conditions, a site that scores as "moderately disturbed" is less certain to have an O/E score truly outside the reference distribution, and repeated sampling at the site along with additional habitat and water quality assessments are recommended (Hubler 2008).

Biological Indices

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

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

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

Methods

Sampling Sites

Ten sites along Whychus Creek were sampled in 2005 and 2009, 13 sites were sampled in 2011, and 11 sites were sampled in 2012. Duplicate samples were taken at one to two sites each year for quality assurance purposes. The sites selected are historic water quality monitoring stations where physical, chemical, and/or biological data has been collected previously (Table 1), and where temperature dataloggers are currently situated. Eight of the ten sites sampled in 2005 were re-sampled in 2009 at or very near the same river mile location; two of the ten sites sampled in 2009 were sampled in the same general area as in 2005 but a different river mile location (i.e. RM 0.5 and RM 3 in 2005 versus RM 1.5 in 2009; RM 23.5 in 2005 vs. RM 24.25 in 2009). All sites sampled in 2009 were re-sampled in 2011, and additional sites were added near both the source and the mouth of the creek. In 2012, two of the new upstream sites added in 2011 were omitted (RM26.5 and RM27), after watershed council staff decided that they did not provide information that was critical and relevant at the watershed scale. The reach at RM30.25 was not sampled in 2012 because it was determined that the river in that region was too deep, wide and fast for standard wadeable stream sampling techniques to be appropriate, and samples taken using kicknets would not be truly comparable to samples from other sites. Overall, sampling sites are distributed broadly along Whychus Creek into downstream (RM 0.5- 9.5), mid-stream (RM 18-19.5), and upstream reaches (RM 23.5 - 30.25). A map of the sampling sites is shown in Appendix A.

Volunteer Training & Sampling Techniques

<u>Training</u>

Volunteer training and stream sampling was done in the same index period across all years: August 20 in 2005 and 2011, August 21 in 2009, and August 18 in 2012. Upper Deschutes Watershed Council staff and volunteers assembled at City Park in Sisters, OR and were trained by Xerces staff in the standardized macroinvertebrate monitoring protocols established for Oregon's wadeable streams (OWEB, 2003). The sampling protocol was demonstrated and each item on the datasheet was explained (see Appendix B for data sheet). A handout with detailed step-by-step sampling instructions was provided, as well as a field guide to Northwest stream macroinvertebrates (Adams *et al*. 2003) and to Northwest freshwater mussels (Nedeau *et al*. 2009), although volunteers were not expected to identify the organisms they collected. The group was then divided into teams of two to four people, and each team received the following equipment: D-frame kick net with 500 µm mesh, metal 500 µm sieve, a square of 500 µm mesh Nitex membrane to line the sieve (in 2012 only), plastic squirt bottle, forceps, thermometer, fiberglass tape measure, 10-gallon plastic bucket, hand lens, 1-liter Nalgene sample jars, 80% ethanol, datasheets, jar labels, and clipboard.

Table 1. Whychus Creek sampling sites, 2005-2012

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, 2012
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, 2012
WC00875[d]	RM 8.75, Rimrock Ranch d/s	44.391278, -121.406182	2011, 2012
WC00900	RM 9, Rimrock Ranch	44.384198, -121.407892	2005, 2009, 2011, 2012
WC00950	RM 9.5, Rimrock Ranch u/s	44.371534, -121.415865	2011, 2012
WC01800	RM 18	44.328342, -121.494534	2005
WC01825	RM 18.25, d/s end DBLT property	44.32781, -121.495406	2009, 2011, 2012
WC01850	RM 18.5, DBLT property	44.326601, -121.500229	2009, 2011, 2012
WC01900	RM 19, DBLT property	44.321523, -121.507461	2005, 2009, 2011, 2012
WC01950	RM 19.5, d/s Camp Polk Bridge on DBLT	44.318741, -121.514961	2009, 2011, 2012
WC02350	RM 23.5, Perit Huntington Rd.	44.29066, -121.53064	2005
WC02425[c]	RM 24.25, City Park, d/s gauge	44.287806, -121.544229	2005, 2009, 2011, 2012
WC02600[c]	RM 26, 4606 Rd. footbridge	44.2730592, -121.555297	2005, 2009, 2011, 2012
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

[a] a duplicate sample was taken at this site in 2005 for quality control
[b] a duplicate sample was taken at this site in 2009 for quality control
[c] a duplicate sample was taken at this site in 2011 for quality control
[d] a duplicate sample was taken at this site in 2012 for quality control

Stream sampling & sample processing

Macroinvertebrate samples were collected from riffle habitat at each site according to standardized protocols (OWEB 2003). Sampling reaches were calculated as 40 times the average wetted width of the stream at the desired sampling point. In 2005 and 2009, each volunteer team calculated the wetted width and paced out the sampling reach length themselves; in 2011 and 2012, this process was greatly streamlined, as watershed council staff calculated wetted widths and clearly flagged the upstream and downstream extent of each sampling reach prior to sampling.

Eight randomly selected riffle habitat areas were sampled within each stream reach. Each sample was collected from a one-foot by one-foot substrate area using a 500 μm D-frame kick net. In reaches that had fewer than eight riffles, two kicknet samples were taken in each of four riffles within the reach. Large rocks and debris in the sampling area were first rinsed into the net to dislodge and collect any clinging organisms and set aside. The substrate was then disturbed using a boot heel or brush handle to a depth of ~10 cm for approximately 30 seconds. The eight individual net samples at each site were placed in a single bucket and large debris was rinsed and removed, and any fish or amphibians were carefully removed and replaced in the stream. Sample material was then concentrated by being poured through a sieve and the composited material was placed into 1-liter Nalgene jars with 80% ethanol added as a preservative. In 2012, the sieves were lined with 500 μm Nitex membrane; this flexible membrane can be lifted from the sieve and the collated sample material rinsed more easily and completely into the sample jar. If the sample contained excessive amounts of sand and gravel, it was elutriated by adding water to the sample bucket, swirling to allow the lighter organic material, including macroinvertebrates, to be suspended above the sand and gravel, then collecting the suspended material on the sieve. After two to three such rinses, the organic material was placed in sample jars separate from the mineral material, to prevent the organisms from being ground up during transport, but all sample material from each site was retained and subsequently examined and picked.

Jars were filled no more than halfway with sample material to ensure preservation. The ethanol in each jar was replaced with fresh ethanol within 48 hours to maintain an 80% concentration, as water leaches from the initial sample material and dilutes the preservative. A simple physical habitat assessment was done at each site to provide data on human use and landscape alterations, substrate composition, water temperature and appearance, and wetted width and depth at each riffle sampled (Appendix B)..

Macroinvertebrate Identification

Samples collected in 2005 were identified by Aquatic Biology Associates, Inc. (ABA; Corvallis, OR); those collected in 2009, 2011, and 2012 were identified by ABR, Inc. Environmental Research & Services (Forest Grove, OR). 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. In 2005, the target count was attained at all but the most upstream sampling site (RM 30.25), which yielded only 397 organisms after the entire sample was picked. In 2009, the target of 500 organisms was attained for all sampling sites, with anywhere from 3-100% of the sample material picked. In 2011, the target count was attained for only 7 of the 13 sites sampled, with anywhere from 38-90% of the sample material picked. Interestingly, the 500-organism count was attained in all but one of the sites spanning RM 1.5 to RM 19, while all sites from RM 19.5 to RM 30.25 yielded only 145 to 385 organisms after the entire sample was picked. In 2012, the target count was attained for eight of the 11 samples collected, with anywhere from 27-100% of the sample material picked. In contrast to 2011, all sites from RM9.5 to RM26 yielded the target count except the RM18.25 sample, which came close to the target number (475 organisms).

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

Data Analysis

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

Most disturbed: O/E = $\leq$0.78
Moderately disturbed: O/E = 0.79 – 0.92
Least disturbed: O/E = 0.93 – 1.23
Enriched: O/E = >1.23

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

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

Biological condition at each site was also assessed using the Oregon Level 3 multimetric Index of Biological Integrity (IBI; OWEB 2003). Individual metrics were calculated and a total IBI score and corresponding stream condition was determined for each site. Metrics include macroinvertebrate community attributes such as taxa diversity; number of sensitive taxa, especially the sensitive stoneflies, mayflies and caddisflies; numbers of tolerant and sensitive taxa; and modified Hilsenhoff Biotic Index (MHBI, a measure of pollution tolerance; Hilsenhoff 1987).

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 data to investigate macroinvertebrate community similarity between sites and across years. Community similarity was also subjected to ordination by non-metric multidimensional scaling (MDS) to investigate site clustering based on sampling year and stream reach location.

Results and Discussion

PREDATOR analysis

Site test results

The site test results file associated with PREDATOR analysis in all years indicated that all predictor variables for the test samples were within the experience of the WC+CB model. Thus, PREDATOR results should be considered valid.

Site O/E

PREDATOR scores showed a general improvement in biological condition from 2005 to 2009. In 2011 most sites either had a similar or lower biological condition compared to 2009, and scores in 2012 were comparable to those seen in 2011 at each site (Table 2 & Figure 1). Observed/expected scores at sites sampled in 2005 rated four sites as most disturbed, four sites as moderately disturbed, and two sites as least disturbed, with one of these sites just slightly above the transition from least disturbed to enriched. In 2009, PREDATOR analysis ranked only one site as most disturbed, five sites as moderately disturbed, and three as least disturbed. In 2011, only one of the 13 sites sampled had an O/E score indicating most disturbed, while five sites were moderately disturbed and three were least disturbed. In 2012, half of the 10 sampling sites received PREDATOR O/E score indicating moderate disturbance, and the remainder were most disturbed.

A comparison of PREDATOR scores across all four sampling years based on the overall stream reach in which the sampling site is located reveals reach-specific differences in condition (Figure 1). Sampling sites in the downstream reaches (RM 0.5 to 9.5) showed the greatest stability in O/E scores. The mean PREDATOR score for downstream reaches in 2005 indicated most disturbed conditions; however, the mean PREDATOR scores in all subsequent years were significantly higher, indicating moderate disturbance, and stabilized after this initial change, as none of the mean scores in 2009, 2011, or 2012 differed significantly from each other (Figure 1B). Similarly, although there appeared to be a downward trend in O/E scores among sites in the upstream reaches, with some individual sites that ranked as moderately disturbed in 2005 and/or 2009 falling to most disturbed, the mean O/E scores for the upstream reaches across all years were not significantly different from each other (Figure 1B).

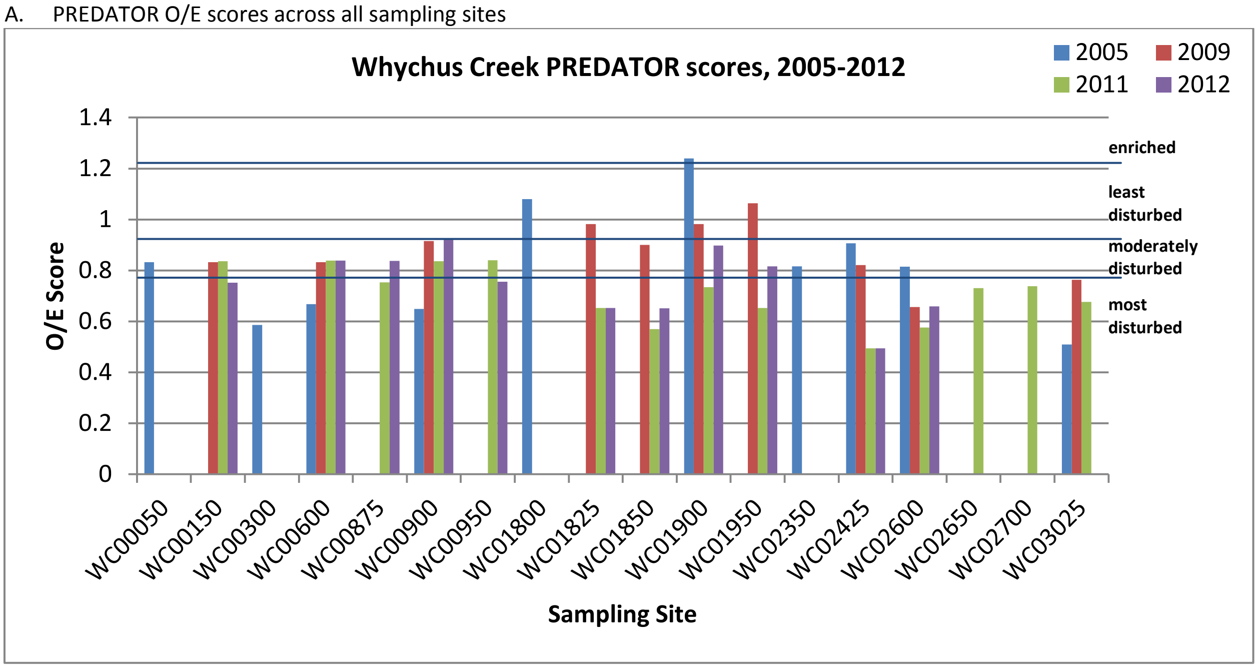
In contrast, the mean PREDATOR score among sites in the midstream reaches (RM 18-19.5) fluctuated more over time, with a high mean score indicating least disturbed conditions dropping significantly by 2011 to a score of most disturbed. The mean PREDATOR score among mid-reach sampling sites was higher in 2012 than in 2011 and came close to the transition between most and moderate disturbance, although the mean O/E scores in 2011 and 2012 were not significantly different. PREDATOR scores in the mid-stream reaches may be expected to be in flux, as sampling sites in this region include the Camp Polk Meadow area. Considering that the stream was only re-directed into the new restored channel in February 2012, it is striking that the mean O/E score for mid-reach sites was actually higher in 2012 than in 2011, although not significantly so. PREDATOR scores in this region will probably continue to fluctuate over the next few years. In addition, although the PREDATOR scores in the upstream reaches did not differ significantly between any years, there appears to be a downward trend that may be significant in the future.

Table 2. PREDATOR O/E Scores. Colored cells indicate conditions of enrichment (orange), least disturbance (green), moderate disturbance (blue), or most disturbed (red).

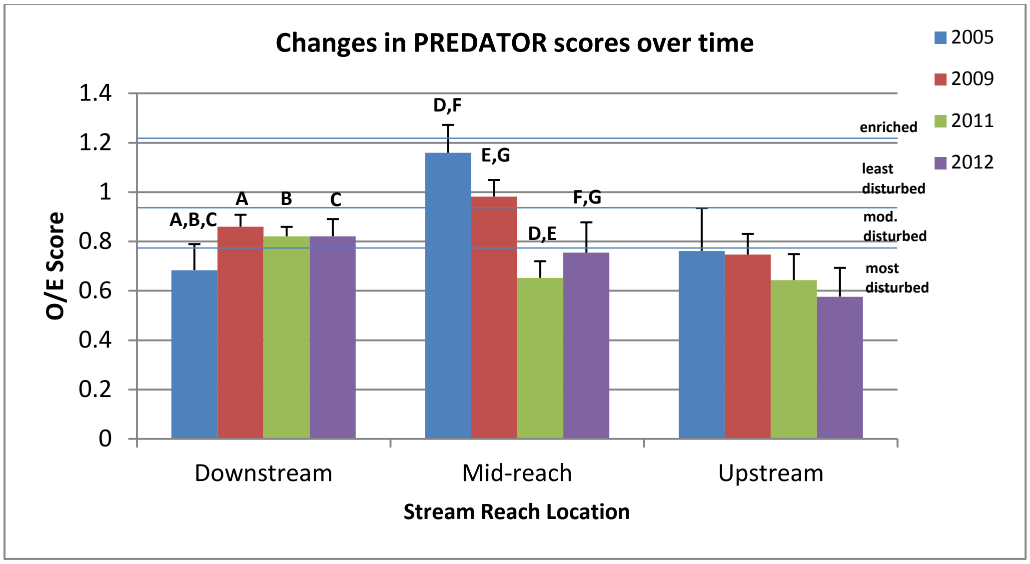
Site	2005	2009	2011	2012
WC00050	0.832696	N/A	N/A	N/A
WC00150	N/A	0.832498	0.836044	0.75244
WC00300	0.585774	N/A	N/A	N/A
WC00600	0.668248	0.831966	0.838606	0.838606
WC00875	N/A	N/A	0.753175	0.836861
WC00900	0.648243	0.91514	0.836575	0.920232
WC00950	N/A	N/A	0.840275	0.756247
WC01800	1.079496	N/A	N/A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

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

A. PREDATOR O/E scores across all sampling sites



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



Missing and replacement taxa

The PREDATOR model creates a matrix comparing the probability of capture of each taxon at each sampling site with the number of sites where that taxon was actually found. Some expected taxa may be absent (missing taxa), while others may be present at a greater number of sampling sites than predicted by the model (replacement taxa). There was a great deal of similarity among missing taxa from 2005 to 2012 when taxa that were absent from $\geq$7 sites are considered (anywhere from nine to 13 sites were sampled in different years, so consideration of taxa that were missing or replacement groups at 7 or more sites represents the majority of sampling sites and is thus a stream-wide phenomenon). In 2005 and 2009, taxa that were missing from $\geq$7 sites where they were expected to occur were almost identical, and included *Epeorus* (a sensitive flatheaded mayfly genus), *Calineuria* (a moderately sensitive perlid stonefly genus), Tanypodinae (a common non-biting midge group), Pisidiidae (common and widespread fingernail clams), and *Malenka* (a common small brown stonefly); additionally, in 2005 Leptophlebiidae (a moderately sensitive prong gill mayfly family) were absent from $\geq$7 of the sites where they were expected to occur. Expected taxa missing from $\geq$7 sampling sites in 2011 included most of the above (Leptophlebiidae, *Malenka*, Pisidiidae, Tanypodinae, and *Calineuria*), as well as Chironomidae (a family of common non-biting midges, of which Tanypodinae is a subfamily), *Optioservus* and *Zaitzevia* (tolerant riffle beetle genera), and *Hydropsyche* (a tolerant net-spinning caddisfly). In 2012, the number of taxa missing from $\geq$7 expected sampling sites was a more restricted subset of that seen in 2011, consisting only of *Malenka*, Tanypodinae, Pisidiidae, *Calineuria*, and *Hydropsyche*, and Chironominae. Chironominae, the only missing taxon group in 2012 that had not been missing in the previous years, is a subfamily of the non-biting midges (Chironomidae), which was among the missing tax groups in 2011.

Substantial similarity was also seen among replacement taxa found at $\geq$7 sampling sites across the four years of sampling. Diamesinae (a non-biting midge group), *Serratella* (a commonly-collected genus of spiny crawler mayfly), *Rhithrogena* (a common and abundant flatheaded mayfly genus), *Acentrella* (a common small minnow mayfly genus), *Narpus* (a common, moderately tolerant riffle beetle genus), and *Atherix* (a common, tolerant watersnipe fly genus) were present as replacement taxa in all years, with the exception of Diamesinae, which was not seen as a replacement group in 2012. Additional replacement taxa seen in 2011 included Nematoda (common roundworms), Turbellaria (flatworms), *Drunella* (a sensitive spiny crawler mayfly genus), *Rickera* (a sensitive stripetail stonefly genus), and *Brachycentrus* (a sensitive genus of humpless case-making caddisfly). The number of replacement taxa seen in 2012 was higher than in previous years, and it differed more in the groups that were over-represented among sampling sites; in addition to having five of the six groups mentioned above as being present in all years, Capniidae (a sensitive family of stoneflies), *Suwallia* (a sensitive genus of chloroperlid stonefly), *Agapetus* (a sensitive genus of saddle case-making caddisfly), *Brachycentrus* (also a replacement taxon in 2011), and *Neoplasta* (a moderately tolerant genus of dance fly) were seen as replacement taxa at ≥7 sites. This may suggest a period of greater dynamism in the macroinvertebrate community, which would not be unexpected, as the creek was moved to its restored channel in Camp Polk Meadow only six months before sampling was done in 2012.

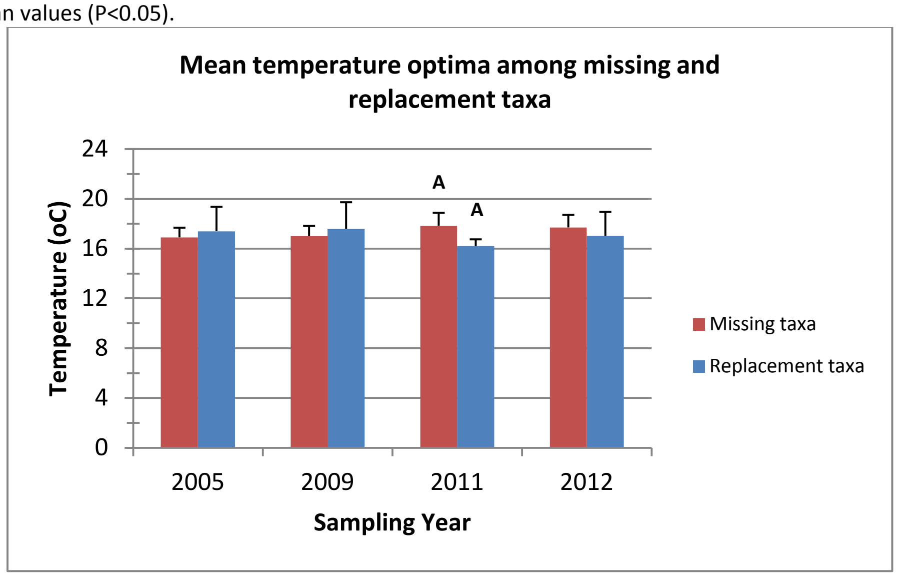
Sediment and temperature optima

A potential explanation for variances from the expected macroinvertebrate community composition was sought by examining differences in sediment and temperature tolerances among missing and replacement taxa. Oregon DEQ developed a set of optima values for specific macroinvertebrate taxa for both seasonal maximum temperature and percent fine sediments (Huff *et al*. 2006) that can be used to assess whether missing or replacement taxa among sampling sites share a range of optima. In each sampling year, differences in mean sediment optima among missing and replacement taxa were far more pronounced than differences in temperature optima (Figure 2). A significant difference in temperature optima between missing and replacement taxa was seen only in 2011, with replacement taxa having significantly lower mean temperature optima than missing taxa. In contrast, the mean sediment optima for replacement taxa was significantly lower than that of missing taxa across all years except in 2009, when the difference between missing and replacement taxa was substantial but not quite significant ($p = 0.0559$).

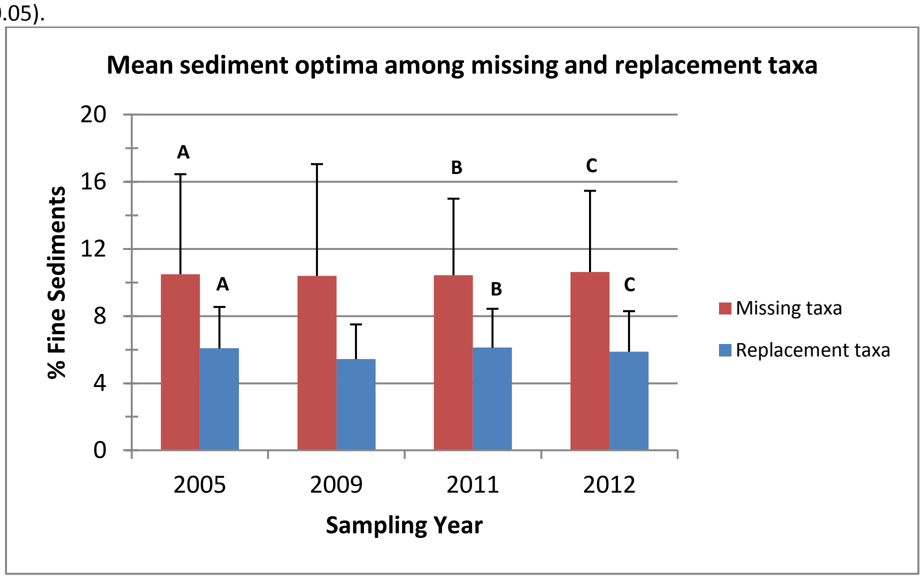
Recent studies have shown that instream restoration work such as channel reconstruction can alter substrate mobility and heterogeneity and affect invertebrate community composition (Albertson *et al*. 2011). Multiple restoration projects done on Whychus Creek to increase flow and restore the stream to its natural channel would be expected to affect both temperature and sediment characteristics, but sediment is a strong and consistent driver of missing vs. replacement taxa, while temperature does not appear to be as significant.

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

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



B. Mean sediment optima of missing vs. replacement taxa. Letters indicate significant differences between mean values (P<0.05).



Multimetric assessment

The OWEB Level 3 stream IBI (genus and species level assessment) consists of 10 metrics. The raw value of each metric is calculated and accorded a corresponding scaled score of 5, 3, or 1, with higher scores indicating better biological condition. Individual metrics are below; the first number shows the raw data range possible for each metric; the number in parentheses is the corresponding scaled IBI score:

- Taxa richness (# of taxa at site): >35 (5), 19-35 (3), <19 (1)
- Ephemeroptera (mayfly) richness: >8 (5), 4-8 (3), <4 (1)
- Plecoptera (stonefly) richness: >5 (5), 3-5 (3), <3 (1)
- Trichoptera (caddisfly) richness: >4 (5), 2-4 (3), <2 (1)

- Number of sensitive taxa: >4 (5), 2-4 (3), <2 (1)
- Number of sediment-sensitive taxa: >2 (5), 1 (3), 0 (1)
- % dominance of the top taxon: <20 (5), 20-40 (3), >40 (1)
- % tolerant taxa: <15 (5), 15-45 (3), >45 (1)
- % sediment-tolerant taxa: <10 (5), 10-25 (3), >25 (1)
- Modified Hilsenhoff Biotic Index (MHBI): <4.0 (5), 4-5 (3), >5.0 (1)

Scaled values for individual metrics are summed to yield a single IBI score for each site, which can reflect a biological condition of minimal (IBI >39), slight (IBI 30-39), moderate (IBI 20-29), or severe impairment (score <20). Overall, IBI scores indicated better biotic conditions than did PREDATOR scores for the same sites. No sites scored as severely impaired in any year; in 2012, two sites received an IBI score indicating minimal disturbance and the remainder scored as being slightly disturbed (Table 3; see Appendix C for individual metric raw data and scaled scores at each site). It was noted in 2011 that the extremely snowy, cold, and wet winter and spring weather conditions may have accounted for the lower scores of four sites that indicated moderate disturbance in 2011 but slight disturbance in 2009 (RM18.5, 19.0, 24.25, and 26.0), and these sites all had substantially higher IBI scores in 2012.

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

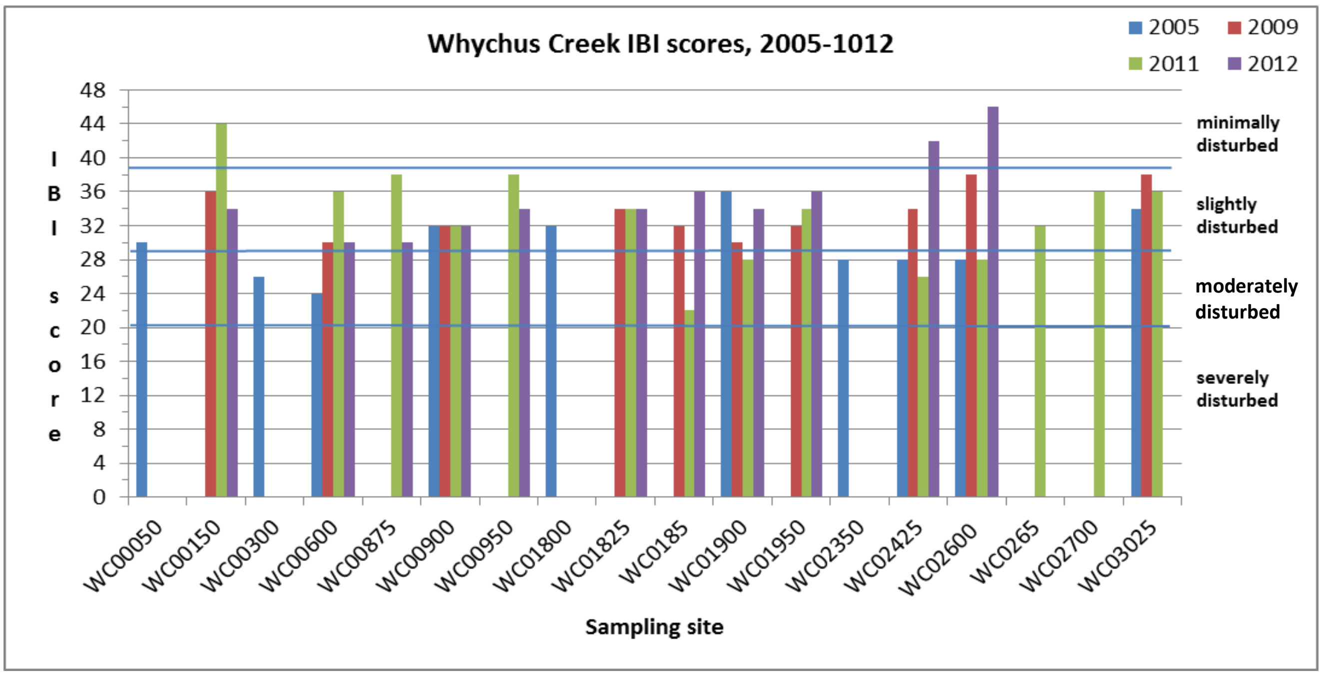
Site	2005	2009	2011	2012
WC00050	30	N/A	N/A	N/A
WC00150	N/A	38	44	34
WC00300	26			
WC00600	24	32	38	32
WC00875	N/A	N/A	40	30
WC00900	36	34	34	32
WC00950	N/A	N/A	38	34
WC01800	32	N/A	N/A	N/A
WC01825	N/A	36	34	34
WC01850	N/A	34	22	36
WC01900	40	34	28	34
WC01950	N/A	34	34	36
WC02425	28	34	26	42
WC02600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

IBI scores present a slightly different picture of stream biotic health than do PREDATOR scores for the same sites and reaches (Figure 3). Similar to what was seen with PREDATOR scores, IBI scores in downstream reaches across all years reflected a more sustained improvement and stability in biological condition for these sites, although the mean IBI score was significantly lower in 2012 compared to 2011. However, in contrast to the greater fluctuation in mean PREDATOR scores seen for sites in the mid-stream reaches, mean IBI scores for sites in this region appeared more stable; both mean IBI and PREDATOR scores were higher in 2012 than in 2011, though this difference was not significant in either case. Mean IBI scores for the upstream sampling sites (RM 23.5 to 30.25) were significantly higher by 2012, in contrast to the apparent downward trend in this region seen in the mean PREDATOR scores, although there were no significant differences in mean annual PREDATOR scores for upstream sites in any year.

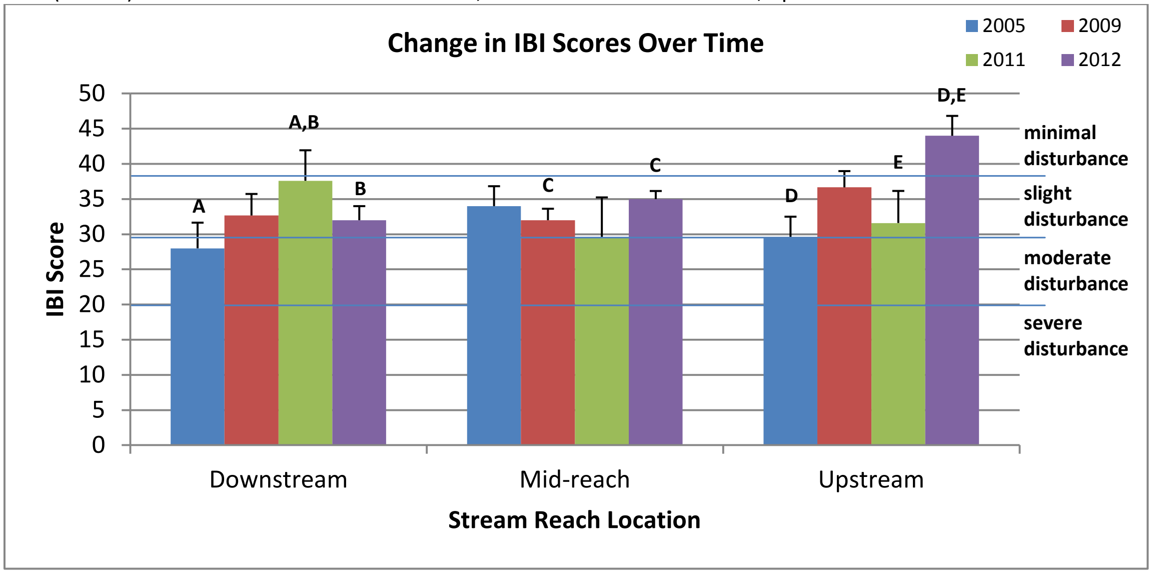
To assess whether particular metrics accounted for the majority of change noted in the scores from year to year, the scaled scores for each individual metric were examined for each site sampled in two or more years (12 sites total; see Appendix C for raw and scaled metric values at each site in each sampling year). While the values for most metrics changed to some extent from year to year at a site, the greatest variation among scaled (scored) values was seen for Trichoptera richness (changed in at least one year at 100% of sites), followed by # of sensitive taxa, % sediment-tolerant taxa, and MHBI (all changed in at least one year at 83% of sites), Ephemeroptera richness , Plecoptera richness, % dominance of top taxon, and % tolerant taxa (all changed in at least one year at 67-75% of the sites). The scaled metrics that remained the most consistent across all sampling years were # of sediment-intolerant taxa and taxa richness, which changed in at least one sampling year at only 50% or 33% of the sites, respectively.

Figure 3. Level 3 IBI scores for Whychus Creek sites

A. IBI scores among all sampling sites



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



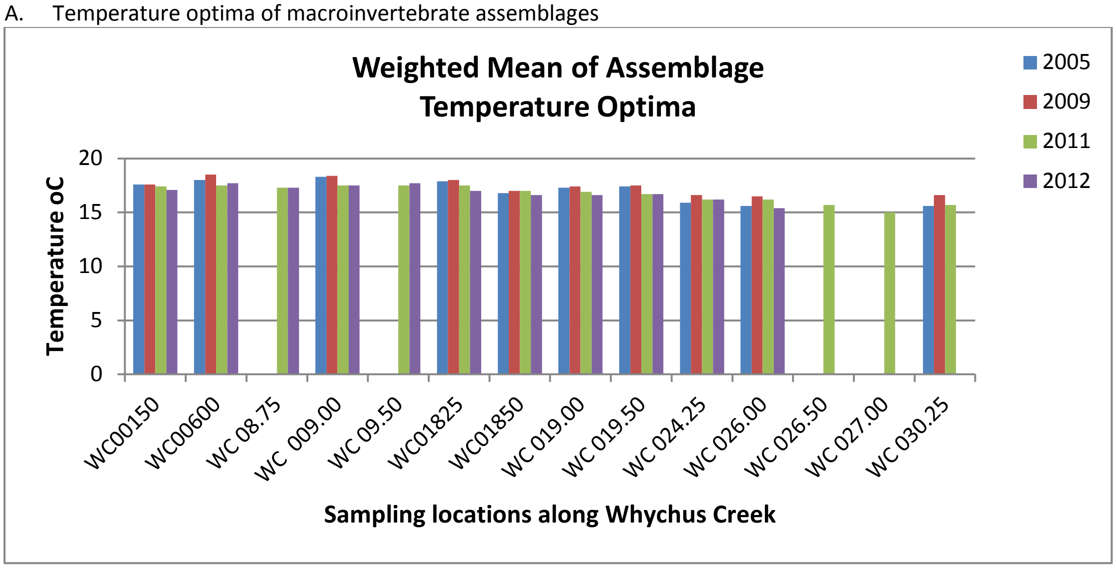
Macroinvertebrate Community Composition

Temperature and Fine Sediment Optima

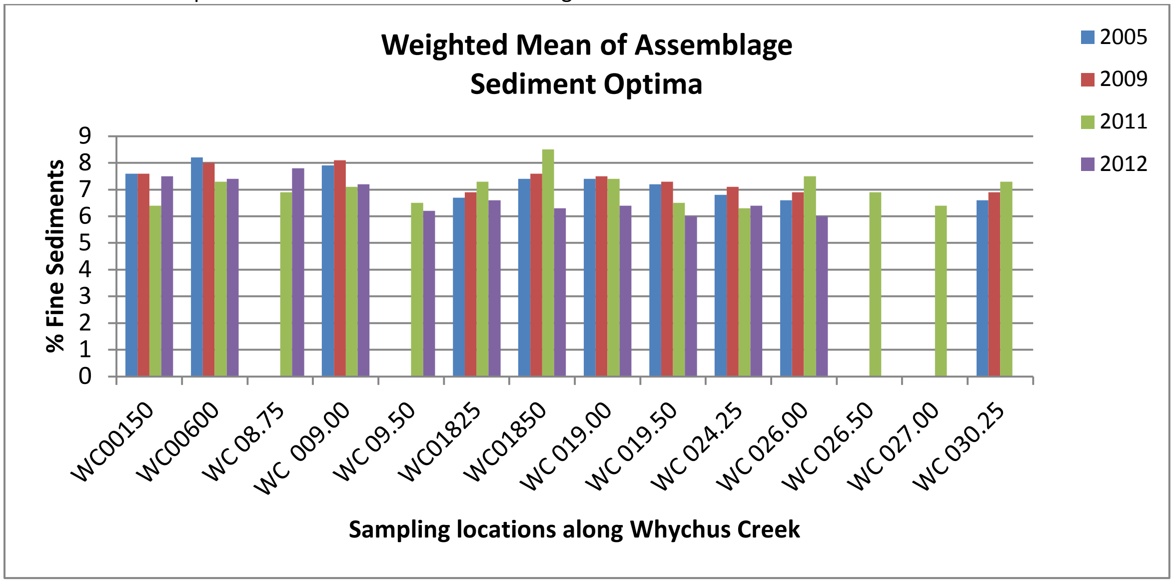
In 2012, data from all collection years were analyzed to determine the mean optima values for temperature and fine sediments exhibited among the macroinvertebrate assemblages at each site over time (Figure 4). The macroinvertebrate communities differed in both temperature and sediment optima from site to site, although the magnitude of change differed among sites, and appeared to vary annually at each site more for sediment than for temperature.

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

A. Temperature optima of macroinvertebrate assemblages



B. Fine sediment optima of macroinvertebrate assemblages

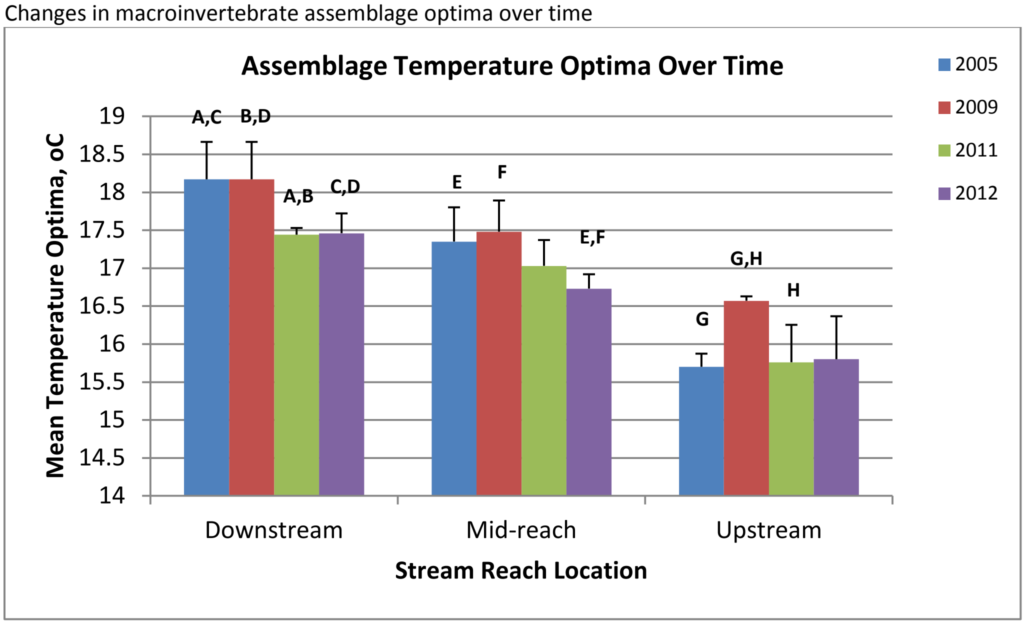


When the data are viewed at the reach level, substantial differences can be seen in overall assemblage values among downstream, mid-reach, and upstream sampling sites, and significant changes in the assemblage optima within individual reaches over time are apparent (Figure 5). The mean temperature optima of the macroinvertebrate assemblages differs overall based on sampling site location, with upstream sampling sites having the lowest overall temperature optima and downstream sites having the highest. This is to be expected based on changes in stream conditions and biota as predicted by the River Continuum Concept (Vannote *et al*. 1980) and is observable in the stream itself, as the colder, faster, deeper headwater region transitions to shallower stretches flowing through areas with a higher proportion of human disturbance and impervious surfaces in the landscape. In addition, a significant downward trend in temperature assemblage optima was seen across time at all three reaches, with the assemblages present in 2005 and 2009 having significantly higher mean temperature optima than assemblages in the same reaches in 2011 and 2012 (Figure 5A). This shift may be stabilizing, as there was no significant difference in mean temperature optima among macroinvertebrate assemblages collected in 2011 and 2012 at sampling sites within the three different reaches.

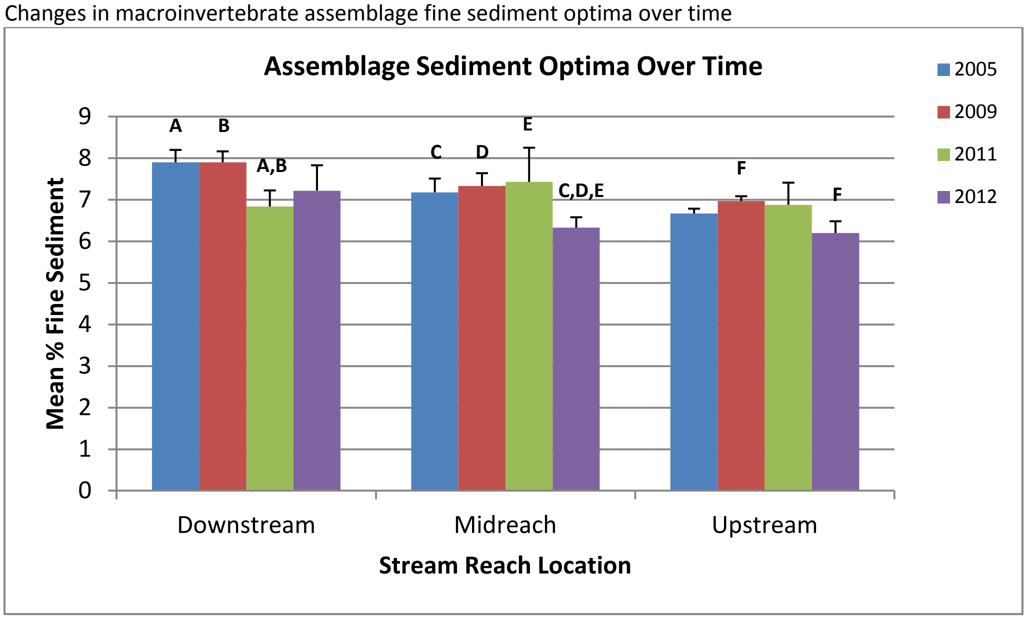
A similar though less pronounced pattern was seen for changes in mean sediment optima among communities within the different reaches (Figure 5B). It is interesting to note that the mean sediment optima for communities in the mid-stream reaches were significantly lower in 2012 compared to all of the three previous sampling years, even though the stream had been diverted into its restored channel at Camp Polk Meadow about six months prior to the 2012 sampling.

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

A. Changes in macroinvertebrate assemblage optima over time



B. Changes in macroinvertebrate assemblage fine sediment optima over time



EPT

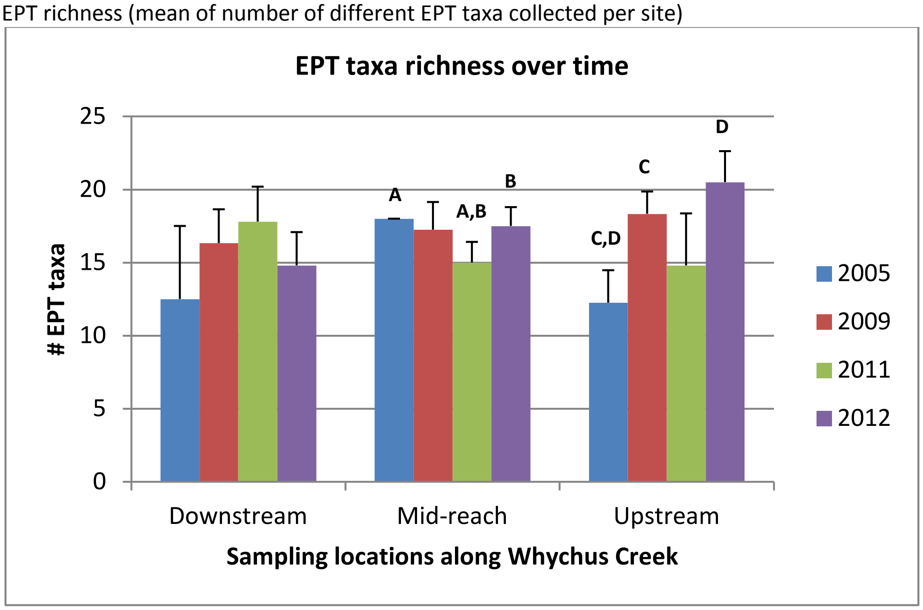
The EPT (mayflies <Ephemeroptera>, stoneflies <Plecoptera>, and caddisflies <Trichoptera>) are considered to comprise the most sensitive groups of aquatic macroinvertebrates and are often used as a measure of stream biological quality. Most taxa in these families require cold, clean, well-oxygenated water, and a greater abundance and diversity of EPT taxa is correlated with better stream conditions. EPT richness and the proportion of all taxa collected comprised of EPT increased slightly with each sampling year. In 2005, 76 taxa were collected across all sampling sites, including a total of 42 EPT taxa (14 Ephemeroptera, 11 Plecoptera, and 17 Trichoptera). Eighty-five taxa were collected among all sites in 2009, including 47 EPT taxa (14 Ephemeroptera, 13 Plecoptera, and 20 Trichoptera). In 2011, 82 taxa were collected across all sampling sites, including 49 EPT taxa (17 Ephemeroptera, 13 Plecoptera, and 19 Trichoptera). In 2012, 79 taxa were collected across all sampling sites, including 44 EPT taxa (19 Ephemeroptera, 11 Plecoptera, and 14 Trichoptera). The proportion of total taxa comprised of EPT was 55.2% in both 2005 and 2009; this increased slightly in 2011 to 59.8% of the total, and dropped to previous levels in 2012 (55.7%).

Because macroinvertebrates respond to aspects of microhabitat that differ among stream reaches, the change in EPT composition across sampling years was examined for sampling sites in the downstream (RM 0.5-9.5), mid-reach (RM 18-19.5), and upstream (RM 23.5-30.25) portions of the creek (Figure 6). The downstream reaches showed a trend toward increased number of EPT taxa, relative proportion of EPT taxa, and relative abundance of EPT from 2005 to 2011, although only the mean relative abundance of EPT among downstream sites was significantly greater in 2011 and 2012 compared to 2009 (Figure 6C). The mean number and proportion of taxa and relative abundance of EPT was similar in mid-reach sites from 2005 to 2011, but was significantly greater in 2012, and mean EPT taxa richness and abundance was higher in 2012 than in any other sampling year in the mid-reach sites.

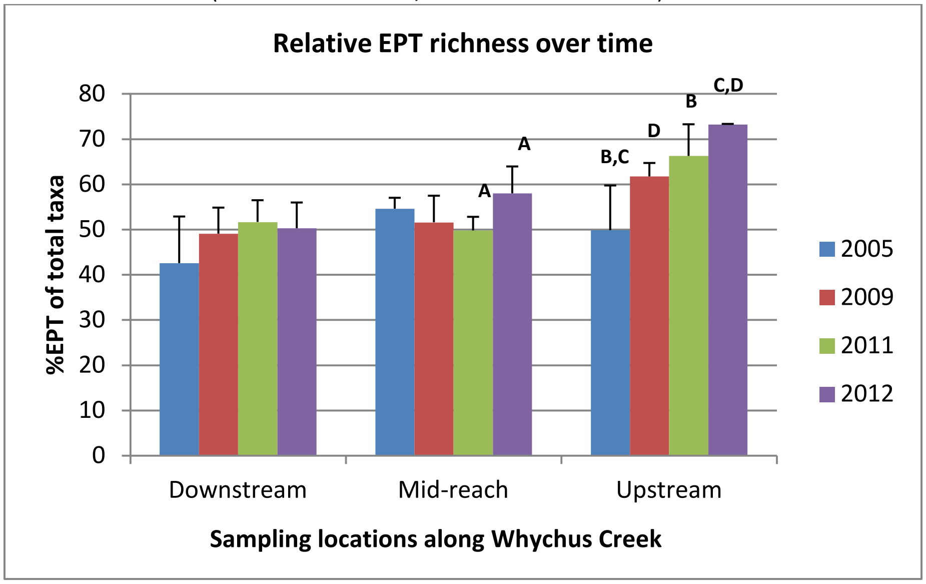
The most sustained and dramatic change in EPT composition was seen among upstream sampling sites, with EPT richness, diversity and relative abundance all trending steadily and significantly upwards. The EPT taxa figure in multiple IBI metrics, and their abundance and diversity is likely one of the driving forces behind the observed increase in mean total IBI scores seen for the upstream reach.

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

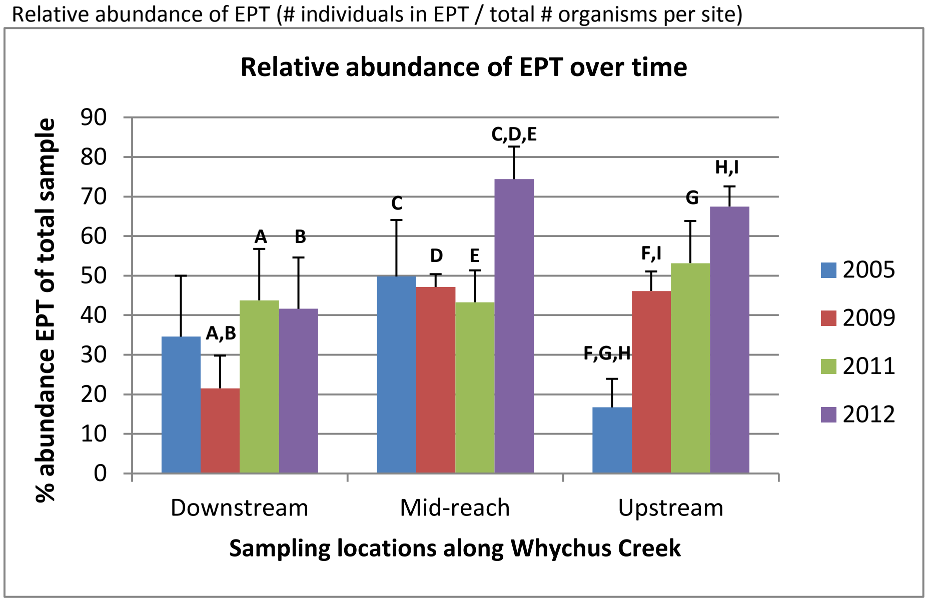
A. EPT richness (mean of number of different EPT taxa collected per site)



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



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



Community composition

In 2005 and 2009, many taxa were found at only a single site (29 taxa and 30 taxa, respectively) and generally with anywhere from one to four individuals. This number was substantially lower in 2011, when only 13 of the 82 total taxa were collected from just one site, also at very low abundance. Similarly in 2012, 18 of the 79 taxa collected were found at only a single site, with no more than one to two individuals. Of the 125 total taxa collected during this project, 36 were present in all four sampling years (see Appendix D for a complete taxa list). These were comprised primarily of riffle beetles (Elmidae), midges (Chironomidae), blackflies (*Simulium*), and small minnow mayflies (Baetidae), all common taxa expected to occur widely and the most abundant and ubiquitous in this study. There were some instances where a difference was seen in the presence or absence of a certain group in 2005 compared to later sampling years. *Rhyacophila* caddisflies were absent from all samples in 2005 but present as multiple different species in 2009-2012, especially in the mid- to upstream reaches of the creek; similarly, the riffle beetle *Lara avara* and several species of *Epeorus* mayflies were absent in 2005 but present in all later years. These groups are more sensitive and prefer cool faster-flowing water. In contrast, several genera of caddisflies in the family Limnephilidae (Northern caddisflies) were present among sites in 2005 but absent from all samples in later years, with the exception of a single individual at one site in 2009, and coenagrionid damselfly nymphs were also seen only in 2005. Coenagrionids are a tolerant damselfly family that prefers to breed in slower waters, and limnephilids can often be found in more lentic (slow-flowing) and/or temporary waters. Thus, changes in stream flow and temperatures following restoration are likely to have influenced this community shift.

Analysis of a Bray-Curtis similarity matrix of square root-transformed data suggests a greater overall change in community composition from 2005 and 2009 compared to 2011 and 2012. Among the six sites that were sampled in all four years (WC00600, WC00900, WC01900, WC02425, WC02600, and WC03025), mean macroinvertebrate community similarity decreased sequentially when the community present in 2005 was compared to that of every subsequent year at the same site. Community composition among all sites sampled across four years was highest for 2005 compared to 2009 (mean similarity = 42.11%), lower for 2005 compared to 2011 (mean similarity = 33.92%) and lowest for 2005 compared to 2012 (30.46%). Similarly, community composition at these sites was more similar in 2011 and 2012 (mean community similarity = 55.68%) than in 2011 and 2009 (mean community similarity = 50.66%). The same pattern was seen among the sites sampled in 2009, 2011, and 2012 (WC01825, WC01850, and WC101950), with mean community similarity highest at the same sites in 2011 and 2012 (mean community similarity = 61.69%) and lowest for the community present in 2009 compared to that at the same sites in 2012 (mean community similarity = 51.89%). The two sites sampled only in 2011 and 2012 (WC08750 and WC00950) had the highest community similarity between the two years (72.03% and 70.08% similarity, respectively), suggesting again that the change in community composition may be slowing and the macroinvertebrate assemblages stabilizing.

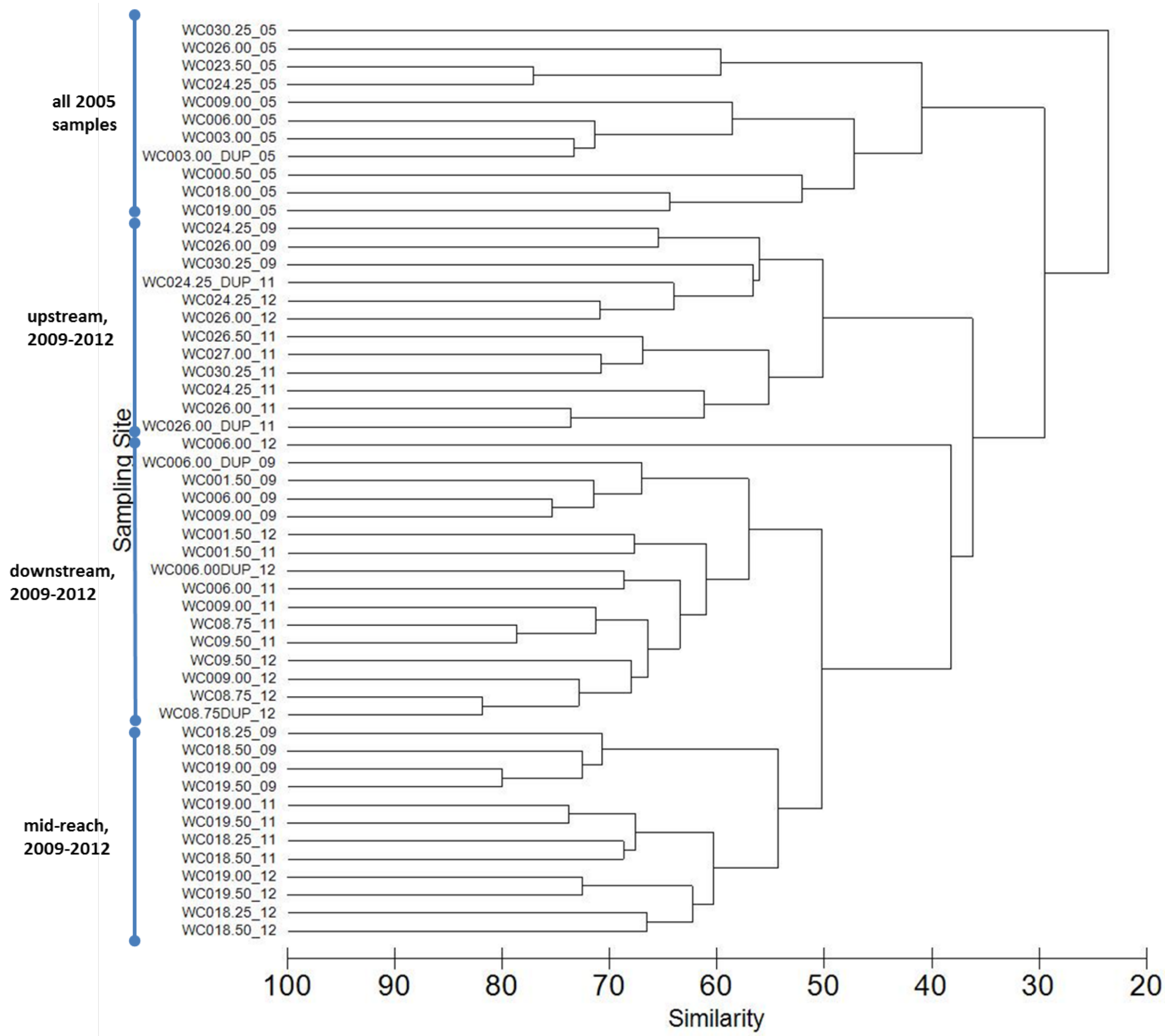
The possibility of a shift in macroinvertebrate community composition followed by greater stability is further suggested by CLUSTER analysis and MDS ordination (PRIMER V6), which grouped all 2005 samples separately from all 2009, 2011, and 2012 samples, with an average similarity between the 2005 sample cluster and the 2009-2012 cluster of only 23.6% (Figure 7A). Samples from other years clustered together based on stream reach location, and within those clusters, communities from samples taken in 2012 were more similar to those taken in 2011 than in 2009. Overall similarity of the communities within each reach group was similar, with all upstream sites from 2009-2012 clustering together at 49% similarity, those from all downstream reaches clustering at 50% similarity, and those from all upstream sampling sites in 2009-2012 clustering together at 53% community similarity (Figure 7A).

MDS ordination of the data illustrates the difference between 2005 and all other sampling years even more dramatically, with the 2005 samples well-separated from all other years. MDS ordination further shows that sample clustering is influenced strongly by site location, with samples taken from similar reaches of the creek (upstream, mid-reach, or downstream) exhibiting the greatest similarity between years (Figure 7B). The greater community similarity of samples taken in 2011 and 2012 compared to 2009 is also apparent for all stream reaches.

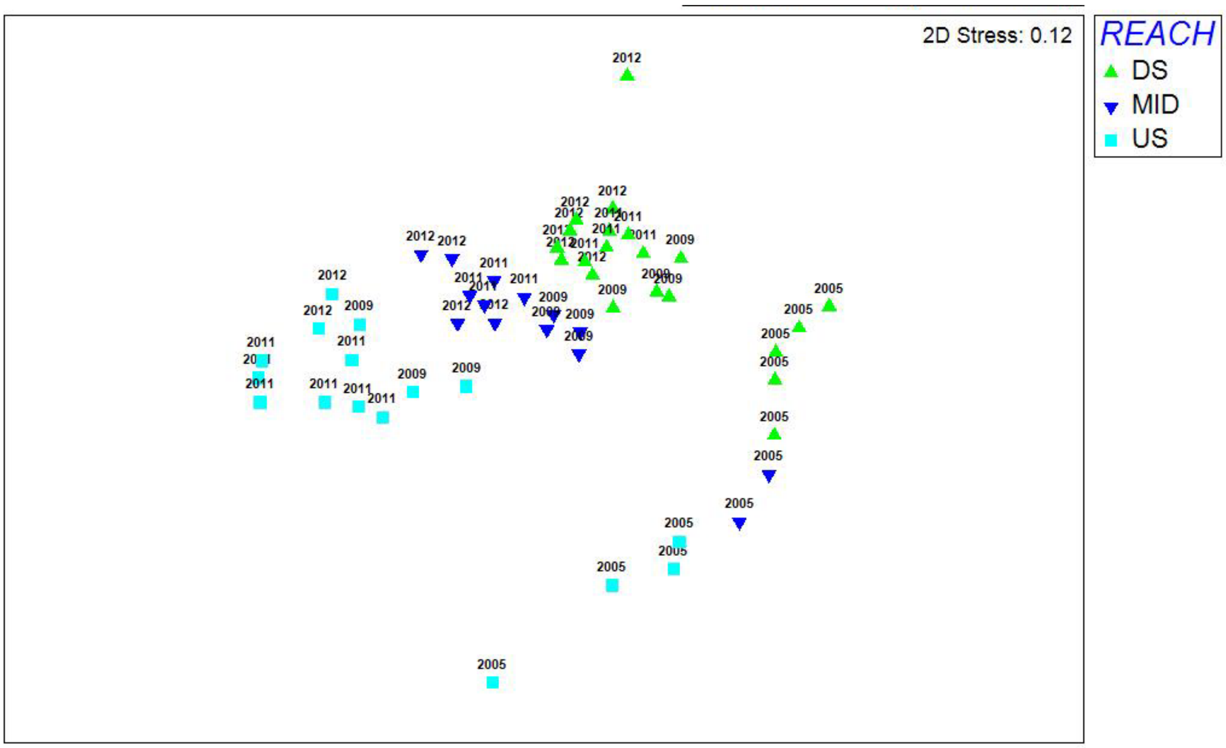
Replicate samples (DUP) taken in all four years for quality assurance purposes clustered most closely with each other in each year, with the exception of one of the two replicate samples taken in 2012 (WC00600), which clustered most closely with the sample taken at the same site the previous year. This indicates that the sampling technique and training provided were sufficiently standardized that volunteers in each year obtained similar results when taking replicate samples in the eight riffles of a selected stream reach. This high level of similarity among each year's duplicate samples makes it more probable that observed differences and similarities in macroinvertebrate community composition between sites and years are real and not an artifact of operator errors.

Figure 7. Similarity in macroinvertebrate community composition according to stream reach location and sampling year.

A. CLUSTER analysis of macroinvertebrate community composition data from 2005-2012. The final two digits of each sampling site label refer to year. DUP indicates a replicate sample taken for quality control purposes.



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



Conclusions

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

Given these caveats, it is essential that any effectiveness monitoring program be conducted on a regular basis over multiple years. Stream degradation occurs over the long-term, and by the same token, recovery of stream biota is not instantaneous. As stream habitat improves, new individuals may be recruited into an area, but the time frame needed for re-colonization and establishment of stable reproducing populations will vary for different taxa with different life histories and colonization abilities, and may be longer than the time frame of many monitoring programs. Only by taking regular "snapshots" of one or more selected biotic communities is it possible to know if the macroinvertebrate community is still changing or if it has stabilized, and whether suspected trends reflect true restoration-related changes in the biotic community or are the isolated result of anomalous conditions in a single year.

Aquatic invertebrate monitoring in Whychus Creek has been done four times within the past seven years, a timeframe that allowed baseline conditions to be established and subsequent post-restoration changes to be identified and assessed. Analysis of macroinvertebrate community data collected from 2005-2012 indicates the following:

- The overall composition of the benthic macroinvertebrate community in Whychus Creek changed substantially from 2005 to 2009, but has shown increasing stabilization from 2009 through 2012. CLUSTER and MDS ordination show that the greatest change in macroinvertebrate community composition occurred between 2005 and 2009, with community composition among each subsequent pair of years becoming increasingly similar. While this could be an artifact of the increased frequency of sampling (i.e. four years between samples taken in 2005 and 2009 versus two years between samples taken in 2009 and 2011), it may also reflect a change in response to restoration activities followed by re-colonization and stabilization within the macroinvertebrate community.

- PREDATOR scores consistently indicate lower biotic conditions than IBI scores at the same sites. Overall, PREDATOR scores indicate that sites in the downstream reach (RM 0.5 to 9.5) showed a slight improvement from 2005 to 2009 that then stabilized across the remaining sampling years. Although a downward trend in PREDATOR score was seen among upstream sites (RM 23.5 to 30.25) from 2005-2012, the change was not significant in any year; however, this may be an early warning of new or increased stressors in this region and should be investigated further. This reach includes the City Park sampling site (RM24.25), which showed a significant decrease in PREDATOR score from 2009 to 2011 and a similarly low score in 2012. City Park is a popular site for camping and recreation and the stream may be suffering in this region. PREDATOR scores at sites within the mid-reach region showed the greatest fluctuation over time, and are likely to continue to change as the stream responds to the recent relocation to its restored channel at Camp Polk.

- IBI scores for the downstream reach sampling sites show a pattern similar to the PREDATOR scores from this reach, with a period of improvement followed by stabilization. Fluctuation among the mid-reach sites was less pronounced, though still present. In contrast to the downward trend in PREDATOR scores seen in upstream sites, however, IBI scores indicate a significant improvement in biotic conditions for sites in this region. The improvement in IBI scores among upstream sampling sites may be accounted for by changes in the sensitive EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa. While EPT richness, relative diversity, and abundance increased over time among all three reach regions, it was most dramatic among the upstream sampling sites. Abundance, richness, and life history characteristic of the EPT factor into multiple IBI metrics, and they may be the driving force behind the observed increase in IBI scores among upstream sites.

- The mean temperature optima of the macroinvertebrate assemblages are consistently higher at downstream sites and lowest at upstream sites. Examination of the missing and replacement taxa at each site as identified by PREDATOR shows that in every year, mean fine sediment optima of replacement taxa is significantly lower than that of the missing taxa. In contrast, differences in mean temperature optima among missing and replacement taxa were significant in only a single year. Stream sediment characteristics thus provide a better explanation for variations in the observed vs. expected macroinvertebrate community than temperature conditions.

- Multiple years of effectiveness monitoring along Whychus Creek have enabled detection of longer term changes and trends. The data suggest that in many parts of the stream, the macroinvertebrate community composition may be stabilizing. However, it would be of interest to conduct an additional year of monitoring to determine whether the community composition among the mid-reach sampling sites continues to fluctuate, and if the trend towards decreased PREDATOR scores among upstream sites persists.

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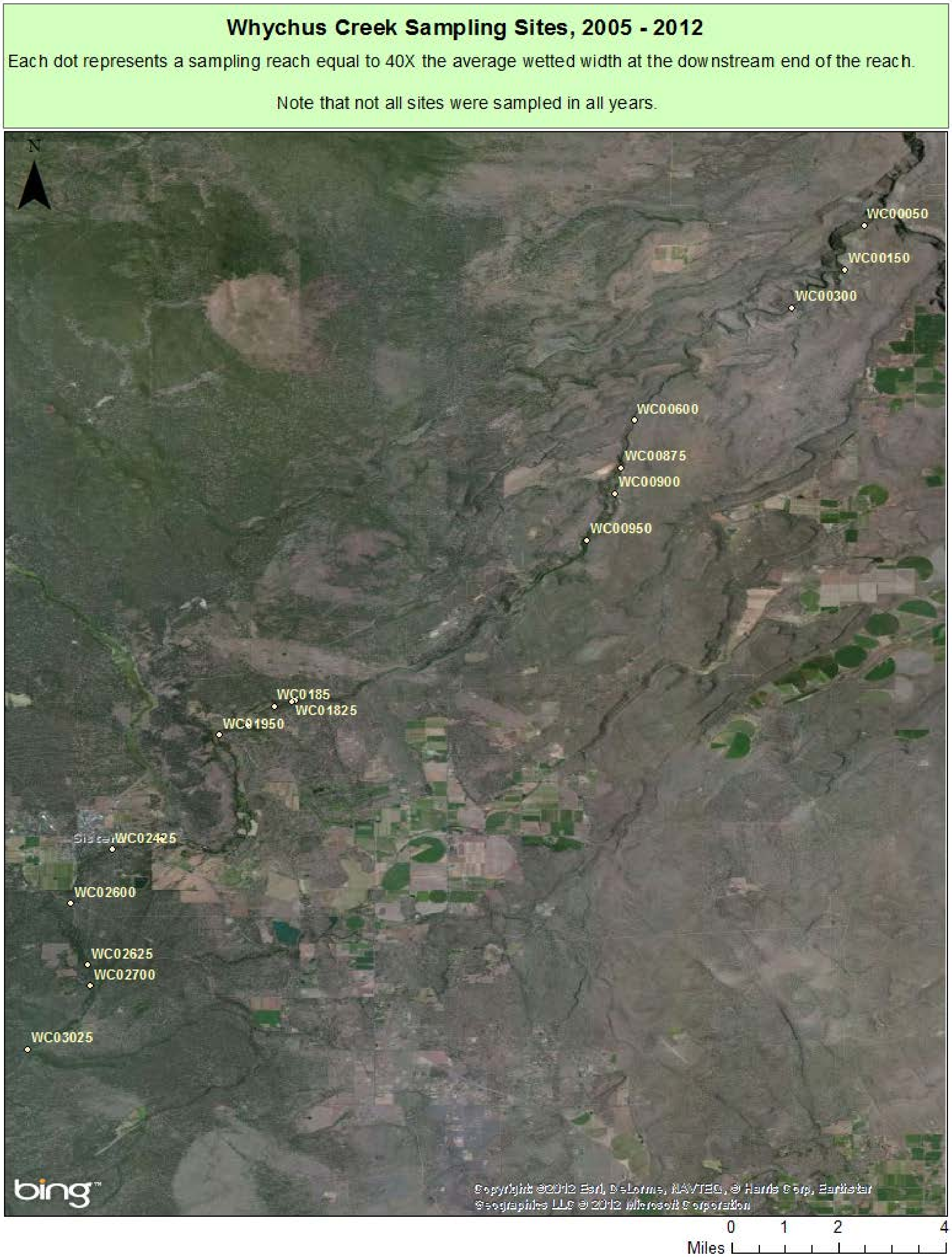
Appendix A. Map of sampling sites along Whychus Creek



Appendix B. Macroinvertebrate monitoring field datasheet

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 ft² OR ___other (describe): total # field duplicate jars _____

Total # sample jars _______

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

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

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

Qualitative observations:

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

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

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

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

<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)

<u>Substrate</u>

% 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)								

<u>Water depth</u>

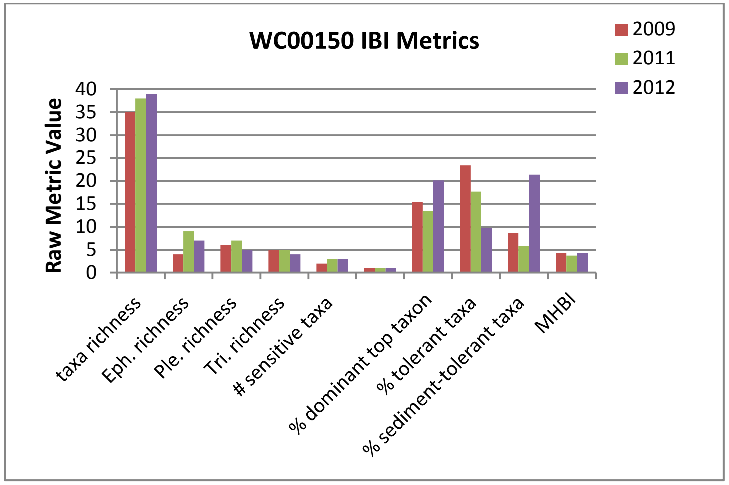
Please record units used (check measuring tape—one side is metric, the other English)

Parameter	Riffle 1	Riffle2	Riffle3	Riffle4	Riffle5	Riffle6	Riffle7	Riffle8
Wetted width								
Depth @ ¼ wetted width								
Depth @ ½ wetted width								
Depth @ ¾ wetted width								

Additional notes or observations (including other wildlife noted):

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

A.



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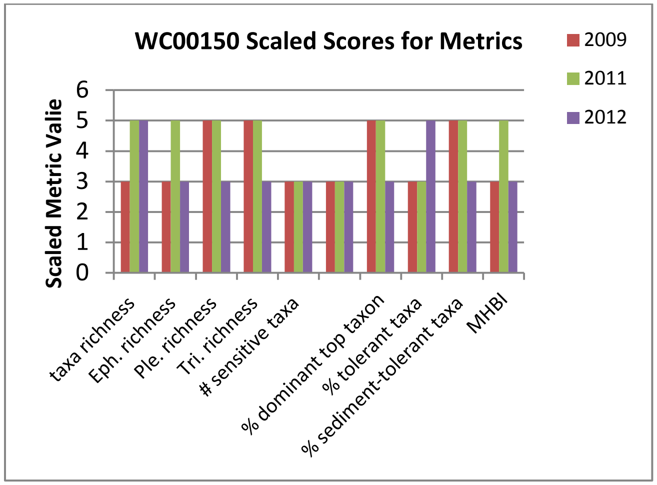

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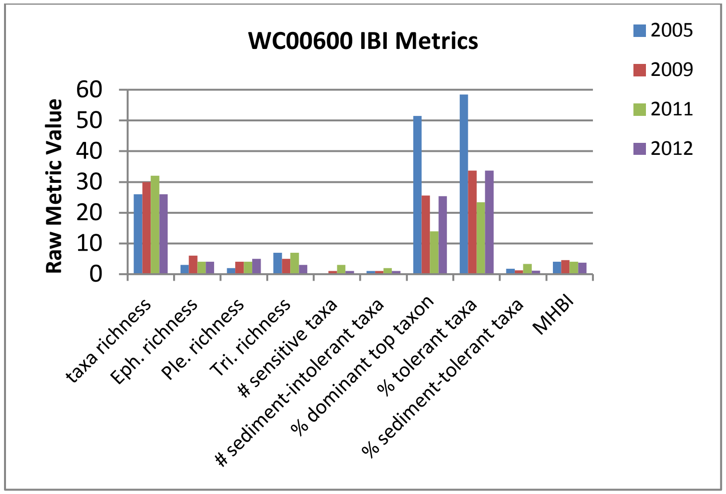

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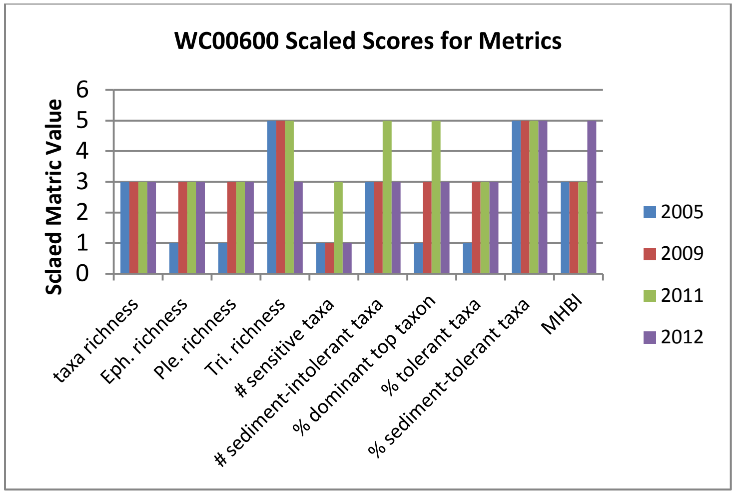

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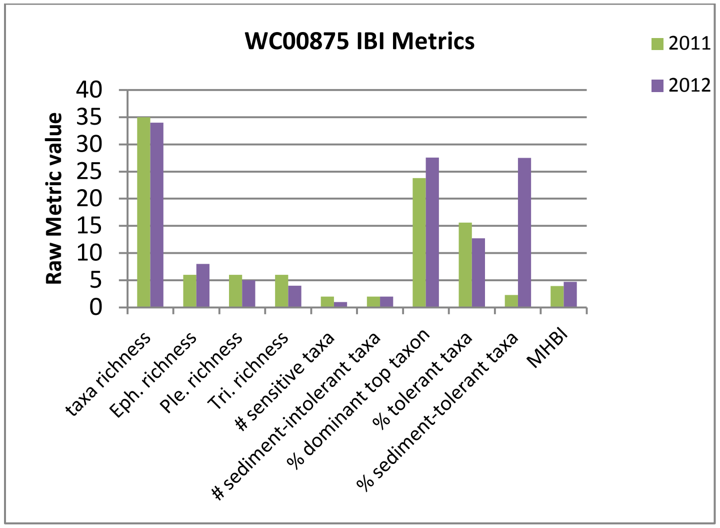
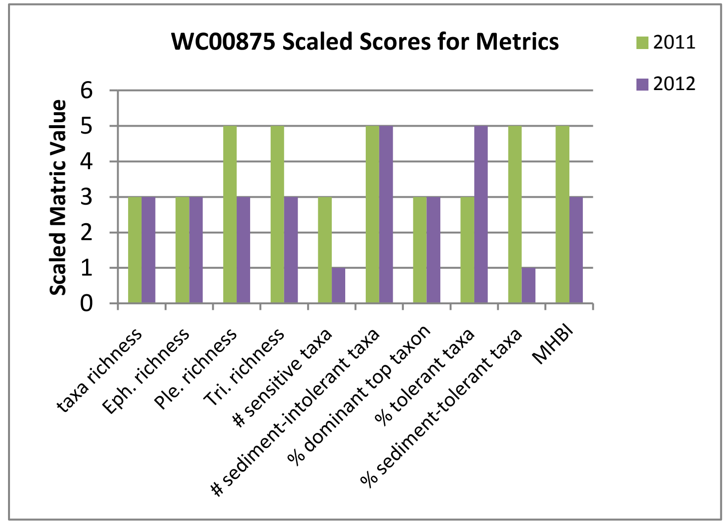
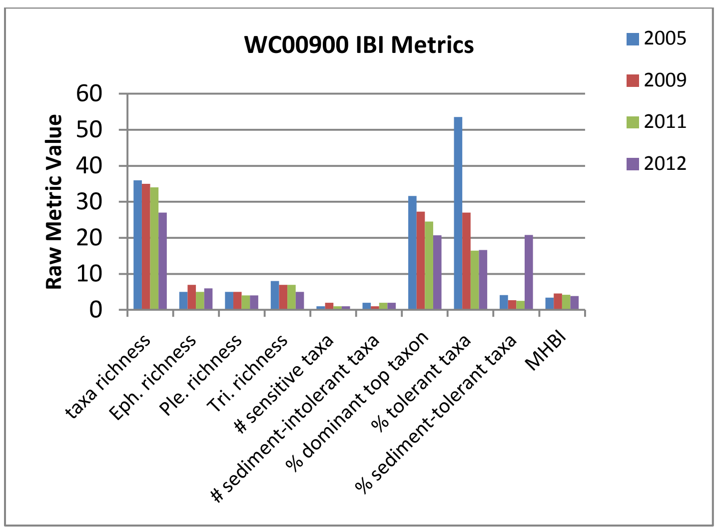

WC00875 IBI Metrics
2011
2012
Raw Metric value
40
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

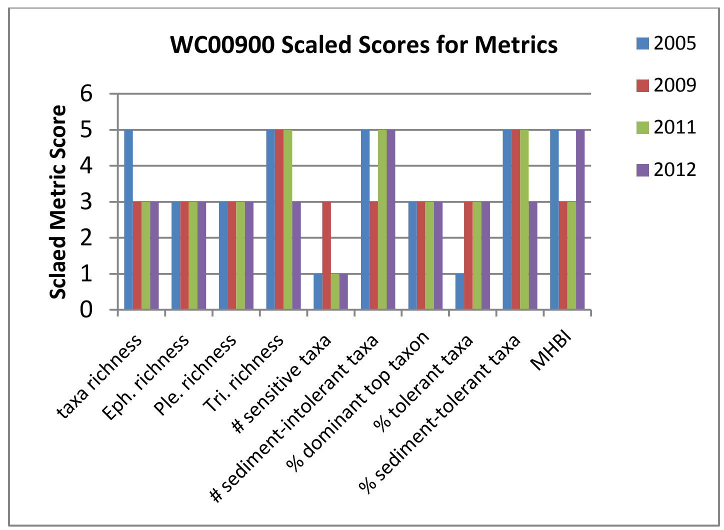
B.


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

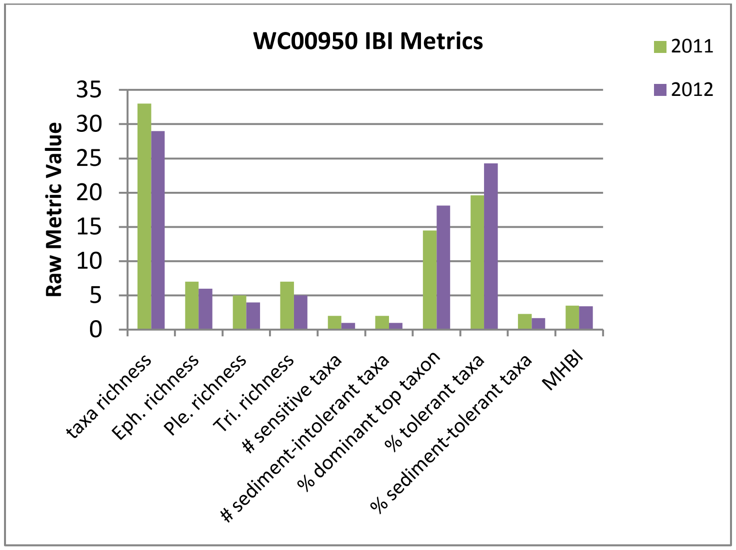
A.


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

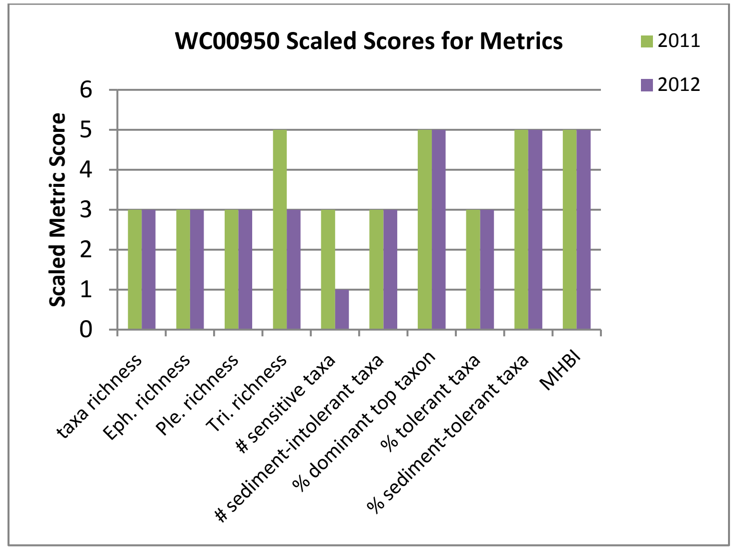
B.


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

A.


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

B.


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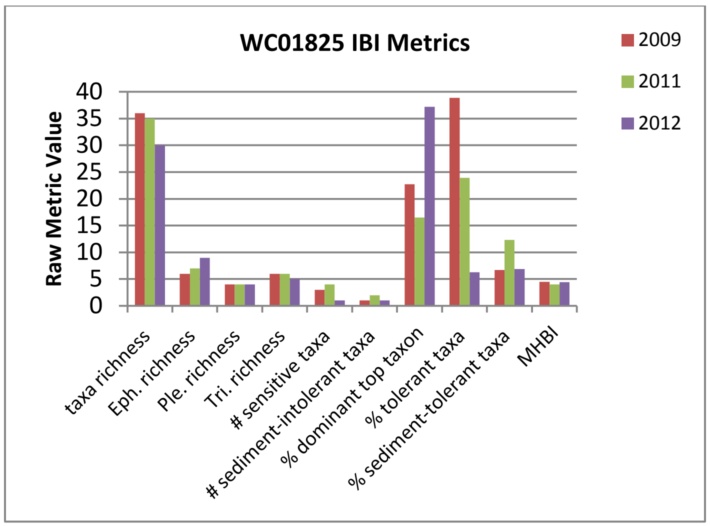
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

A.


WC01825 IBI Metrics
2009
2011
2012
Raw Metric Value
40
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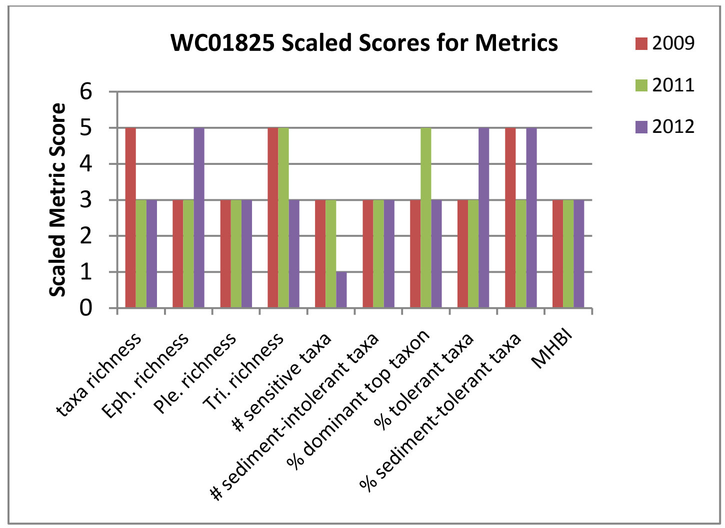
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

B.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2009
2011
2012
Scaled Metric Score
6
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4
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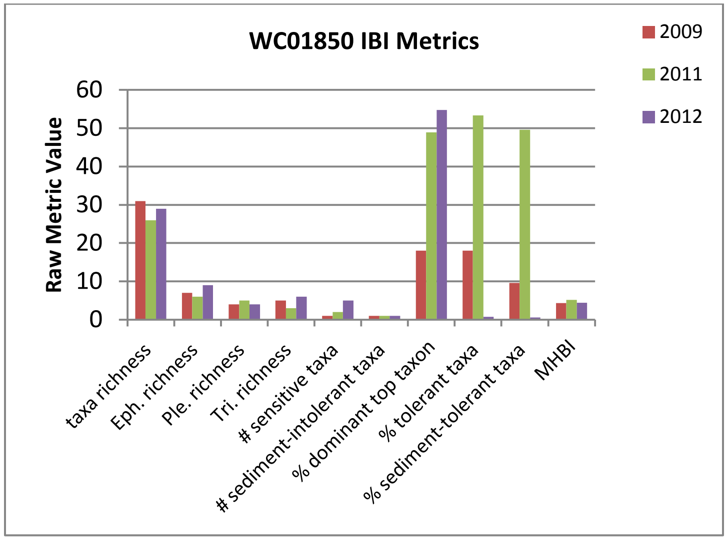
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

A.


WC01850 IBI Metrics
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Raw Metric Value
60
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taxa richness
Eph. richness
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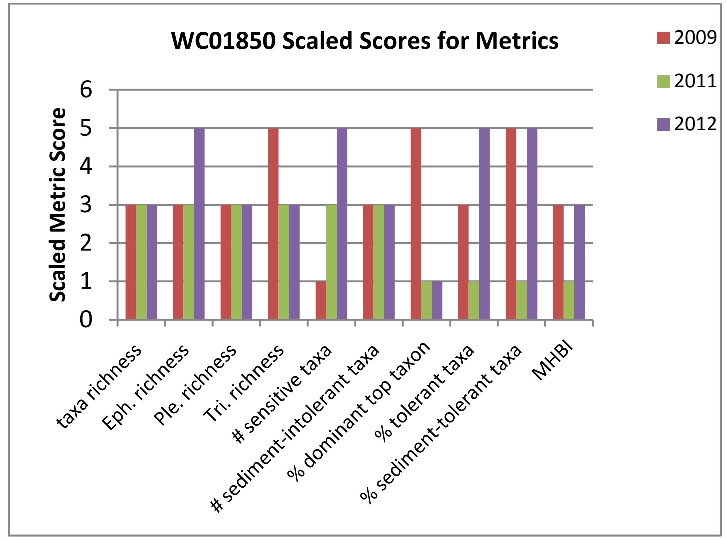
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

B.


WC01850 Scaled Scores for Metrics
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2012
Scaled Metric Score
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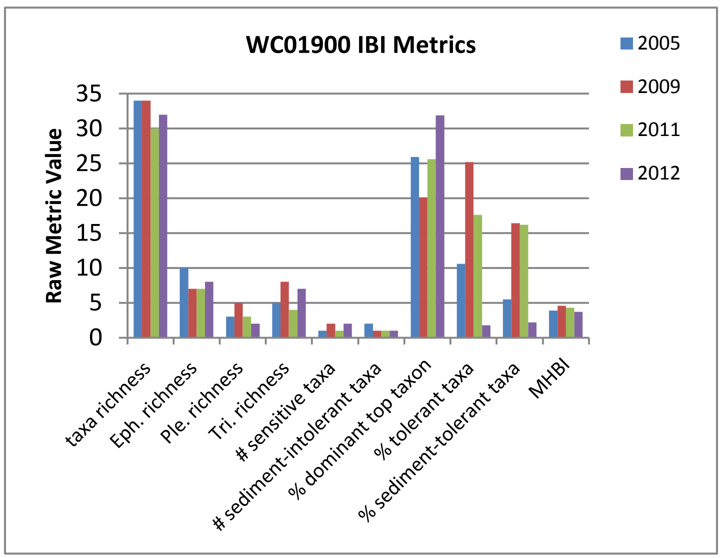
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

A.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2009
2011
2012
Raw Metric Value
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taxa richness
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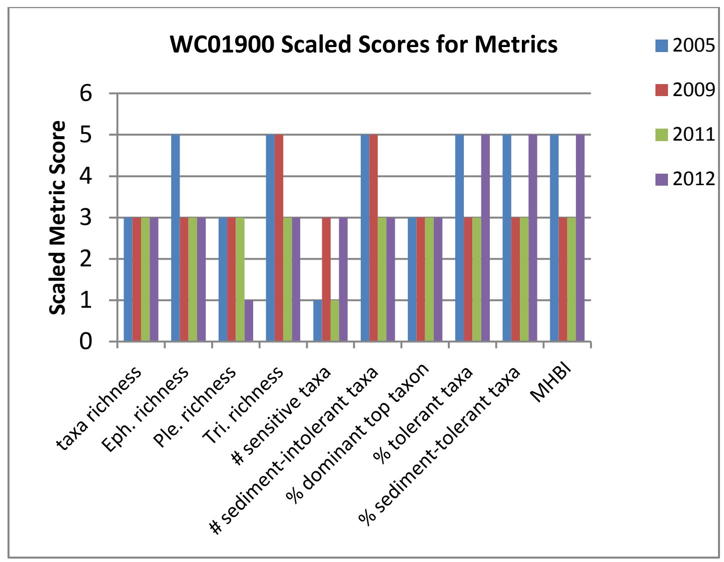
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

B.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2005
2009
2011
2012
Scaled Metric Score
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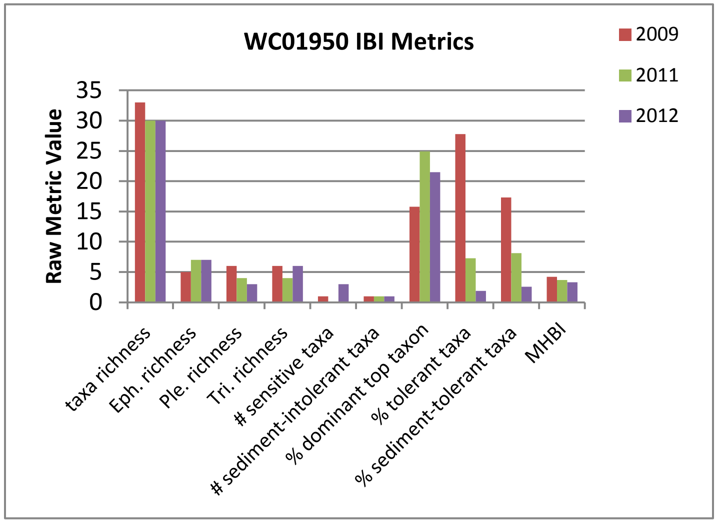
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

A.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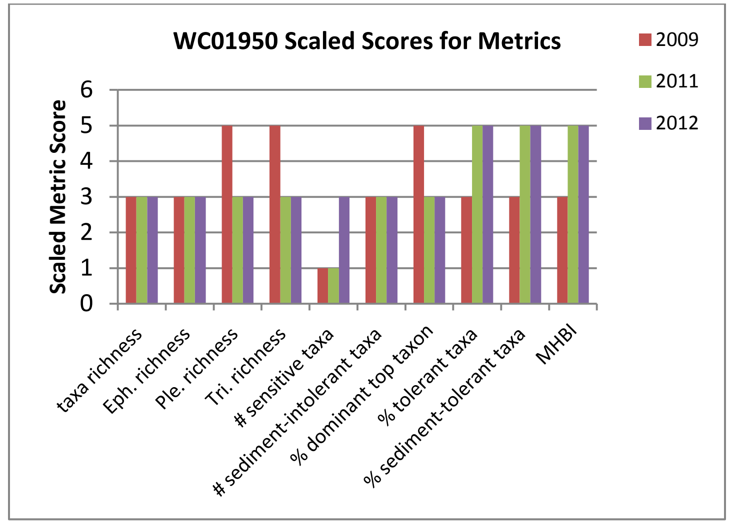
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI
2009
2011
2012

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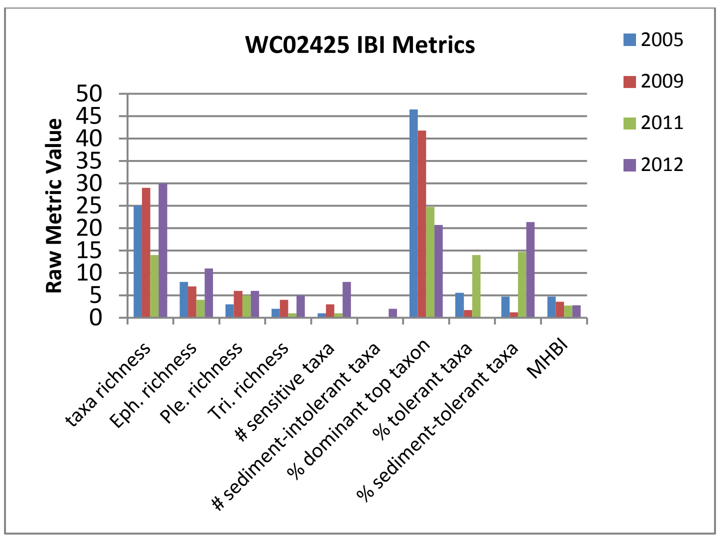
Scaled Metric Score
0 1 2 3 4 5 6
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI
2009
2011
2012

A.


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

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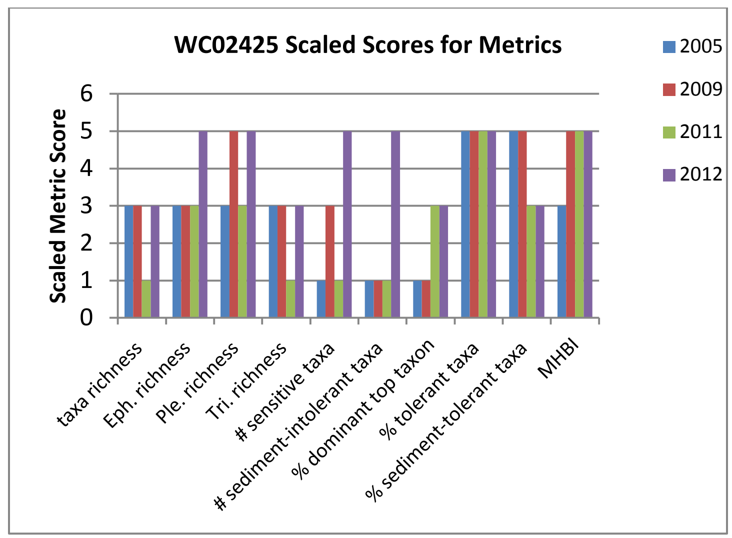
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Scaled Metric Score
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taxa richness
Eph. richness
Ple. richness
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sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI
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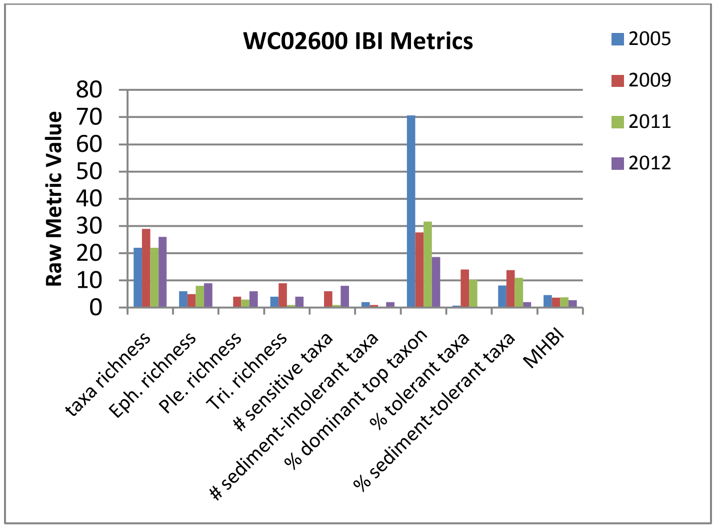
A.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30
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70
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2005
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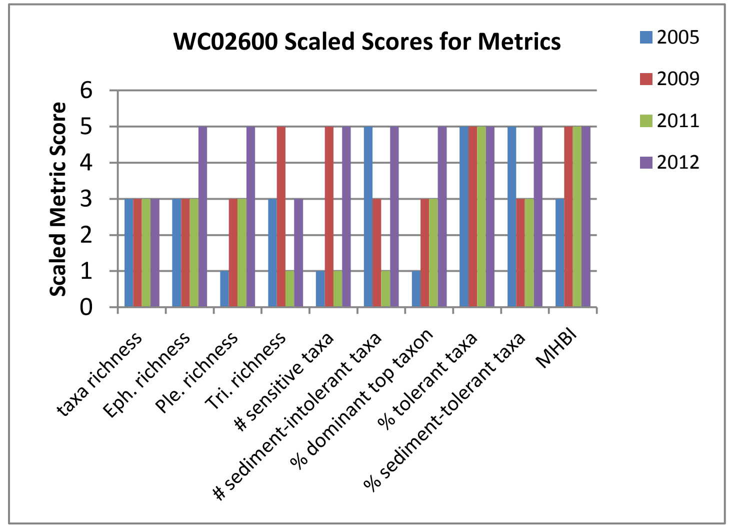
2011
2012
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

B.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Scaled Metric Score
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2005
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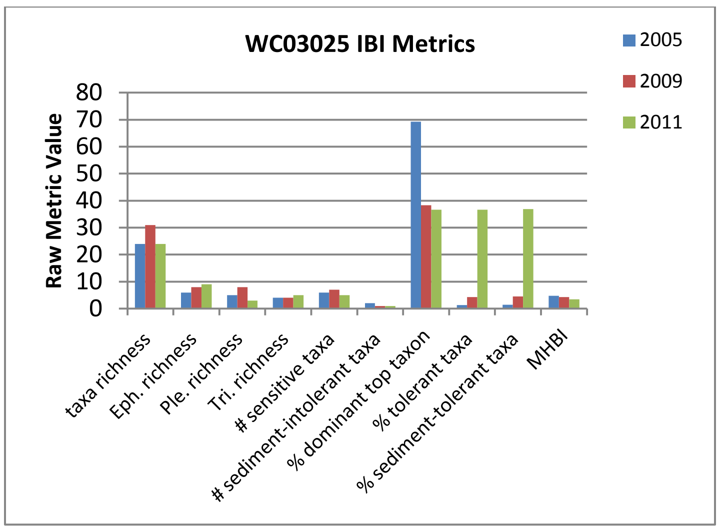
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

A.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Raw Metric Value
0
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70
80
2005
2009
2011
taxa richness
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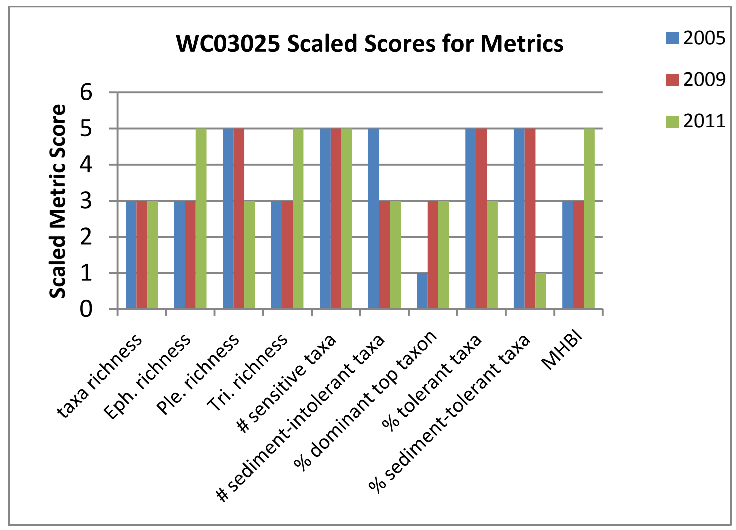
Ple. richness
Tri. richness
sensitive taxa
sediment-intolerant taxa
% dominant top taxon
% tolerant taxa
% sediment-tolerant taxa
MHBI

B.


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

Appendix D. Macroinvertebrate Taxa List for Whychus Creek, 2005-2012

Phylum/subphylum	Class/Subclass	Order	Family	Genus	Species	2005	2009	2011	2012
Platyhelminthes	Turbellaria					√	√	√	√
Annelida	Oligochaeta					√	√	√	√
Nematoda						√	√	√	√
Arthropoda / Crustacea	Malacostraca	Decapoda	Astacidae	Pacifasticus			√		
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			√	√	
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	Wiedemannia		√			
Arthropoda	Insecta	Diptera	Tipulidae	Antocha		√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Cryptolabis		√			
Arthropoda	Insecta	Diptera	Tipulidae	Dicranota			√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Hesperoconopa		√	√		
Arthropoda	Insecta	Diptera	Tipulidae	Hexatoma		√	√	√	√
Arthropoda	Insecta	Diptera	Tipulidae	Limnophila			√	√	
Arthropoda	Insecta	Diptera	Tipulidae	Rhabdomastix			√		
Arthropoda	Insecta	Diptera	Athericidae	Atherix		√	√	√	√
Arthropoda	Insecta	Diptera	Dixidae	Dixa			√		√

Arthropoda	Insecta	Diptera	Chironomidae	Tanypodinae		√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Chironominae		√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Diamesinae		√	√	√	√
Arthropoda	Insecta	Diptera	Chironomidae	Orthocladiinae		√	√	√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Ceratopogoninae			√		
Arthropoda	Insecta	Diptera	Ceratopogonidae	Dasyhelea				√	√
Arthropoda	Insecta	Diptera	Ceratopogonidae	Forcipomyia				√	
Arthropoda	Insecta	Diptera	Blephariceridae	Bibiocephala					√
Arthropoda	Insecta	Diptera	Blephariceridae	Blepharicera		√	√	√	√
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	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	Attenella	margarita				√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	hystrix			√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Serratella		√			
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella		√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella (Serratella)	tibialis		√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	dorothea			√	
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Ephemerella	excrucians		√	√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Caudatella	hystrix	√	√		√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	spinifera	√		√	√
Arthropoda	Insecta	Ephemeroptera	Ephemerellidae	Drunella	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	Cinygma					√
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Cinygmula		√	√	√	√
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	Tricorythodes		√			
Arthropoda	Insecta	Ephemeroptera	Leptophlebiidae	Paraleptophlebia		√	√	√	√
Arthropoda	Insecta	Megaloptera	Sialidae	Sialis		√			
Arthropoda	Insecta	Plecoptera	Perlidae			√	√	√	
Arthropoda	Insecta	Plecoptera	Perlidae	Calineuria	californica	√	√		
Arthropoda	Insecta	Plecoptera	Perlidae	Hesperoperla		√			
Arthropoda	Insecta	Plecoptera	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	Amphinemoura					√
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	Betteni Gr.		√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Brunnea/ Vemna Gr.		√	√	√
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Hyalinata Gr.		√	√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	narvae		√	√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Nevadensis Gr.			√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	grandis		√		
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	Vagrita Gr.		√	√	
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophila	valuma		√	√	√
Arthropoda	Insecta	Trichoptera	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		√			√
Arthropoda	Insecta	Trichoptera	Sericostomatidae	Gumaga					√
Mollusca	Gastropoda	Basommatophora	Ancylidae	Ferrissia		√			
Mollusca	Gastropoda	Basommatophora	Physidae	Physa		√			√
Mollusca	Gastropoda	Neotaenioglossa	Pleuroceridae	Juga			√		√
Mollusca	Gastropoda	Basommatophora	Planorbidae			√			√

Native Fish Monitoring in Whychus Creek

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Abstract

A suite of restoration actions on Whychus Creek aims to restore the stream habitat, flows, fish passage, and water quality necessary to support self-sustaining populations of reintroduced Chinook salmon (*Oncorhyncus tshawytscha*) and steelhead trout (*Oncorhyncus mykiss*), native resident redband trout, and bull trout. Steelhead were reintroduced to Whychus Creek beginning in 2007, with salmon first reintroduced in 2009, and hundreds of thousands of fry and thousands of smolts continue to be released annually. Restoration partners including Portland General Electric and USFS conduct native fish monitoring annually in Whychus Creek to quantify *O. mykiss* density and census *O. mykiss* redds. PGE also traps smolts outmigrating from the tributary arms of Lake Billy Chinook to generate smolt production estimates. The Whychus Creek Monitoring Plan identified fish populations as measured by PGE monitoring data as a biological indicator of restoration effectiveness. However, due to the ongoing and recently completed status of multiple restoration projects, continued annual releases of steelhead and Chinook, and the inability to differentiate between juvenile redband and steelhead pending genetic analysis, available data on fish populations are inadequate to evaluate fish response to restoration. While recognizing this limitation, the Upper Deschutes Watershed Council continues to summarize PGE's fish data for Whychus Creek annually to track the status and trends of fish populations. *O. mykiss* have accounted for the majority of fish caught in every year sampled since 2007. Juvenile O. mykiss population estimates stabilized in 2012 after falling from 2008 to 2011, in 2012 ranging from 7-32 fish/100m^2 in four Whychus reaches. Juvenile spring chinook densities continue to decline despite ongoing annual releases. Only twelve *O. mykiss* redds were detected in 2012: redd counts were rendered ineffective by high turbidity and low visibility and are not considered representative of the extent of *O. mykiss* spawning. Ongoing refinements of PGE native fish monitoring protocols will continue to improve the utility of the resulting data to describe fish populations in Whychus Creek. 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 data will more directly reflect stream habitat and watershed conditions and may provide a more useful indicator of restoration effectiveness.

Introduction

Anadromous populations of summer steelhead (*Oncorhyncus mykiss*) and spring Chinook salmon (*Oncorhyncus tshawytscha*) were extirpated from the Upper Deschutes sub-basin 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 sub-basin. Steelhead fry were reintroduced in Whychus Creek and the Crooked River system in 2007 and have been released in the hundreds of thousands every year since; Chinook fry and smolts, and steelhead smolts, were first released in 2009 (Table 2). Under the 2005 FERC re-licensing agreement for the Pelton-Round Butte hydroelectric project, Portland General Electric (PGE) conducts native fish monitoring annually in Upper Deschutes sub-basin tributaries supporting salmon and steelhead reintroduction, and publishes multiple reports summarizing monitoring results. The primary objective of PGE's native fish monitoring is to describe *O. mykiss* populations and spawning activity within the study reaches, including population size, size-frequency distributions, redd counts, and timing of spawning. PGE additionally monitors juvenile migration from Upper Deschutes tributaries into Lake Billy Chinook to estimate smolt production as well as to establish migration timing and rate and document Chinook and steelhead size and growth. In 2009 USFS also conducted mark-recapture surveys on Whychus Creek to generate population estimates for steelhead and Chinook.

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

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

Kunkel (2010) evaluated the relative utility of fish populations in Whychus Creek as biological indicators of restoration effectiveness. This report identified multiple obstacles to the use of available data to evaluate the short-term response of Whychus fish populations to changes in habitat resulting from restoration actions. Foremost among these are annual releases of steelhead fry and smolts, which at least in part drive *O. mykiss* abundance and mask any response of the existing population to changes in stream conditions. Releases are scheduled to continue until it is determined that steelhead populations may be sufficiently established as to be self-sustaining (ODFW and CTWS 2008). Attempts to estimate abundance of juvenile steelhead versus redband trout are confounded by the inability to differentiate juvenile steelhead and redband without conducting expensive genetic analyses. Short term changes to habitat following restoration frequently do not reflect the fully restored condition of the creek and may even adversely affect fish populations; independent of population fluctuations resulting from ongoing yearly releases, fish population trends may not begin to reflect habitat suitability until years after restoration projects are completed, once sediments, stream banks, 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. Even though we are unable to draw any conclusions about the extent to which changes in stream conditions will support recovery of steelhead and salmon populations in Whychus and the Deschutes over the longer term, information about the current status and trends of fish populations in Whychus may nonetheless provide some measure of interim stream conditions. Stable or increasing fish populations may indicate interim stream conditions that provide suitable habitat and adequate resources to support fish survival, whereas a decreasing trend may flag compromised or unsuitable conditions for fish survival. We anticipate native fish monitoring data will become a useful indicator of restoration effectiveness over the long term, as restored communities achieve full ecological function. PGE will conduct genetic analysis to determine relative proportions of juvenile redband and steelhead five and ten years after returning fish are first passed upstream of the hydroelectric project (2017 and 2022). These data will provide some insight into population dynamics and interactions between the two life histories. In the interim, UDWC will continue 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.

This technical report summarizes results of PGE's 2012 native fish monitoring in Whychus Creek, compiled from PGE reports. We compare *O. mykiss* population estimates and redd counts from 2012 to 2007-2011 results; 2006 native fish monitoring data were collected using different methods and are not comparable to 2007-2012 data and are therefore not considered in this report.

Fish Populations in Whychus Creek

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

Fish species presently occurring in Whychus Creek include native redband trout (*Oncorynchus mykiss*), non-native brown trout (*Salmo trutta*), longnose dace (*Rhinichthys cataractae*), and sculpin (*Cottidae*). Non-native brook trout (*Salvelinus fontinalis*) were caught during native fish monitoring surveys in 2007 and 2008 but have not been observed since; bridgelip sucker (*Catostomus columbianus*) were last observed in 2006, also during PGE's native fish monitoring. No current sampling effort specifically targets either of these species, but they are believed to persist at low abundance in Whychus Creek (M. Hill 2011, personal communication). Native bull trout (*Salvelinus confluentus*) have been observed in Whychus Creek below Alder Springs (Fies *et al* 1996). PGE captured one bull trout each year in the Alder Springs area from 2003-2005 (M. Hill 2009, personal communication) but none have been captured or observed since 2005.

Chinook salmon

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

Sockeye salmon

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

O. mykiss

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

Redband trout and summer steelhead naturally coexist in the lower Deschutes River downstream from the Pelton Round Butte dams. Resident and anadromous forms of *O. mykiss* may have both historically occurred in Whychus Creek as well. It is uncertain to what extent both life history forms will again coexist in Whychus Creek as steelhead runs are reestablished. The habitats of juvenile redband and steelhead are similar, and there will likely be some level of interaction between the two life history forms, including competition for resources and perhaps spawning interaction. Zimmerman and Reeves (1999) provide evidence that steelhead and redband trout in the lower Deschutes River are reproductively isolated by their utilization of different spawning habitats and by differences in their time of spawning. Behnke (2002) also suggests that populations of resident and anadromous forms of *O. mykiss* may maintain their genetic distinction by spawning in separate areas within the same stream system. Conversely, a recent study from the Hood River showed that up to 40% of anadromous steelhead genes in a given generation were from wild redband trout, suggesting extensive inter-breeding between the two life histories (Christie *et al* 2011). Ackerman *et al* (2007) and Cramer and Beamesderfer (2006) suggest that Whychus Creek will produce primarily anadromous, not resident, O. mykiss, based on stream flows and temperature.

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

Methods

O. mykiss and Spring Chinook Juvenile Density

PGE fisheries managers selected four study reaches in 2002 (Figure 1) representative of the range of habitats in Whychus Creek (Lewis 2003). A fifth reach was added in 2009. Reach 1 is located downstream from Alder Springs at river mile (rm) 1.5/river kilometer (rkm) 2.5. Reach 2 is downstream from USFS Road 6360 at rm 6 (rkm 9). Reach 3, at Camp Polk (rm 19/rkm 25.5) was sampled from 2006 through 2011 but was not sampled in 2012 following diversion of the stream from the straightened channel, where prior sampling had occurred, into the restored meadow channel. Reach 4 is downstream from Hwy 20 in Sisters at rm 23.5 (rkm 34.5). Reach 5 is located on Wolftree property at rm 17.5 (rkm 25).

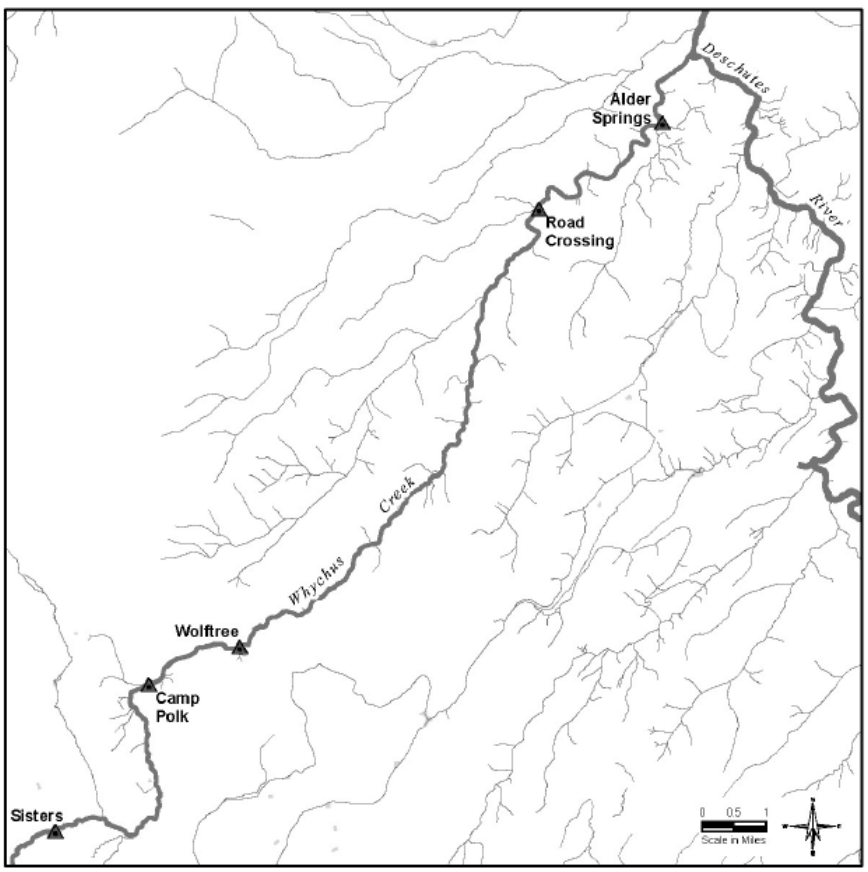

Figure 1.

Study reaches on Whychus Creek for fish population estimates. Alder Springs, Road 6360 Crossing and Sisters reaches (reaches 1-4) have been surveyed annually since 2006; Camp Polk was surveyed from 2006-2011; and Wolftree (Reach 5) has been surveyed annually since 2009. Reproduced with permission from Quesada *et al* 2013**.**

Fish population sampling was conducted during the low flow period from September 5-12, 2012. Study reach lengths ranged from 110-219 m, determined by the location of habitat characteristics allowing the secure placement of blocknets. Where extensive habitat changes had occurred since 2009, stream sections were surveyed following ODFW stream survey protocols (Moore *et al* 2006). Block nets were situated above and below survey sections within each reach, with an additional net placed midsection to evaluate blocknet effectiveness and the mark-recapture sampling assumption of a closed population. As in 2011, high flows in 2012 prevented effective use of blocknets at some sites, and in several reaches sampling was conducted without blocknets. Researchers used natural habitat breaks thought to constrain fish movement, such as riffle/pool breaks, to define the beginning and end of reaches where blocknets proved ineffective. Mark-recapture sampling was conducted from the stream section below the reach to the stream section above to determine whether fish were moving out of the reach. Where any marked fish were recaptured in the stream section below or above the reach, all fish captured in these sections were added to the Peterson population estimate for that reach. Where no marked fish were recaptured in these sections, fish captured in the same sections were not included in the population estimate for that reach, although they were still used for length frequency data analysis.

Mark-recapture electrofishing was conducted following protocols adapted from ODFW (Scheerer *et al* 2007), as described in Quesada *et al* (2013). 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 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.) In 2012 ODFW limited chinook surveys to mark-recapture electrofishing and did not conduct snorkel surveys.

O. mykiss and Spring Chinook Smolt Production

To estimate numbers of steelhead, Chinook and redband juveniles outmigrating from Whychus Creek and other tributaries and mark downstream migrants for later identification, PGE has operated screwtraps annually on the Metolius, Deschutes and Crooked Rivers, their tributaries, and/or respective arms of Lake Billy Chinook. Recovery of marked fish provides data to estimate the relative contribution of smolts from each tributary, and, if measures are taken in the future to differentiate redband and steelhead, will provide information on out-migration of redband. Fish traps deployed in Whychus Creek in 2009 and 2010 were difficult to operate effectively due to vandalism and widely fluctuating flows during the downstream migration period (Hill and Quesada 2010, Hill and Quesada 2011). Resulting 2009 and 2010 data were inadequate to develop smolt production estimates. In 2011 attempts to trap out-migrating smolts in Whychus were abandoned. Instead, in 2011 a screwtrap was located on the Upper Deschutes River Arm of Lake Billy Chinook. In 2012, the screwtrap was replaced with a Merwin trap in the same location (Figure 2). The trap was operated seven days a week throughout the migratory season, from March 8th until May 25th, 2012, and checked daily. Captured fish were recorded by species. Steelhead and Chinook smolts were measured and checked for passive integrated transponder (PIT) tags. Smolts greater than 60 mm and 2.0 g without PIT tags were tagged.

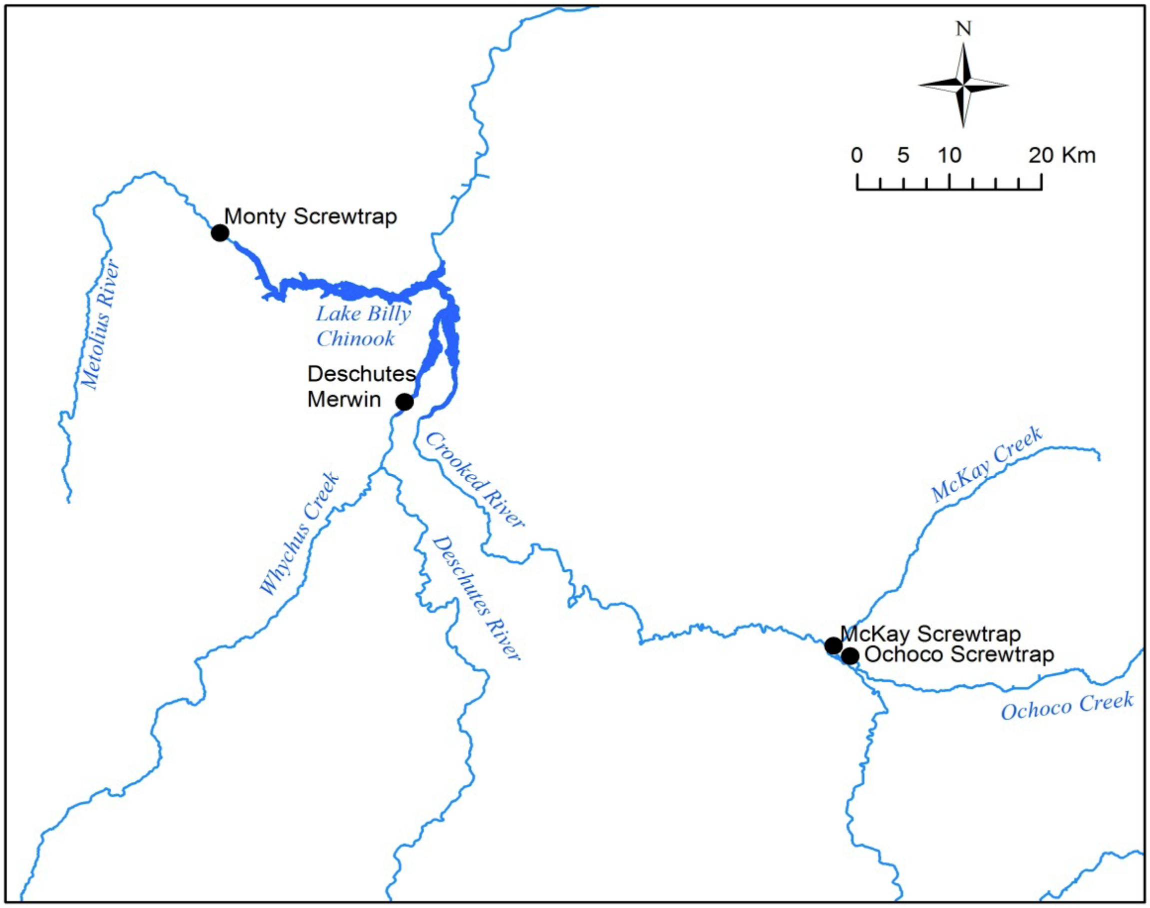

Figure 2.
2012 trap locations. Reproduced with permission from Hill and Quesada 2013.

O. mykiss Redd Counts

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

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

In 2012 PGE conducted redd counts in the two designated index sites at Alder Springs (rkm 2) and immediately upstream of Camp Polk (rkm 27), in two randomly chosen sites at Rimrock Ranch (rkm 14) and at rkm 38 upstream of Sisters, and in the four original reaches surveyed since 2006, at Alder Springs Creek (reach 2), Lewis Woodpecker Creek (reach 3), rkm 25 (reach 6), and rkm 26 (reach 7) (Figure 2).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.

In 2009 and 2010 PGE measured a sub-sample of redband trout redds as baseline data to use in differentiating redband and steelhead redds and spawning distribution once adult steelhead return to Whychus. Measurements of pot and spill included width, length, depth, and substrate size. Temperature data were also collected. PGE did not measure redband redds in 2011 nor in 2012.

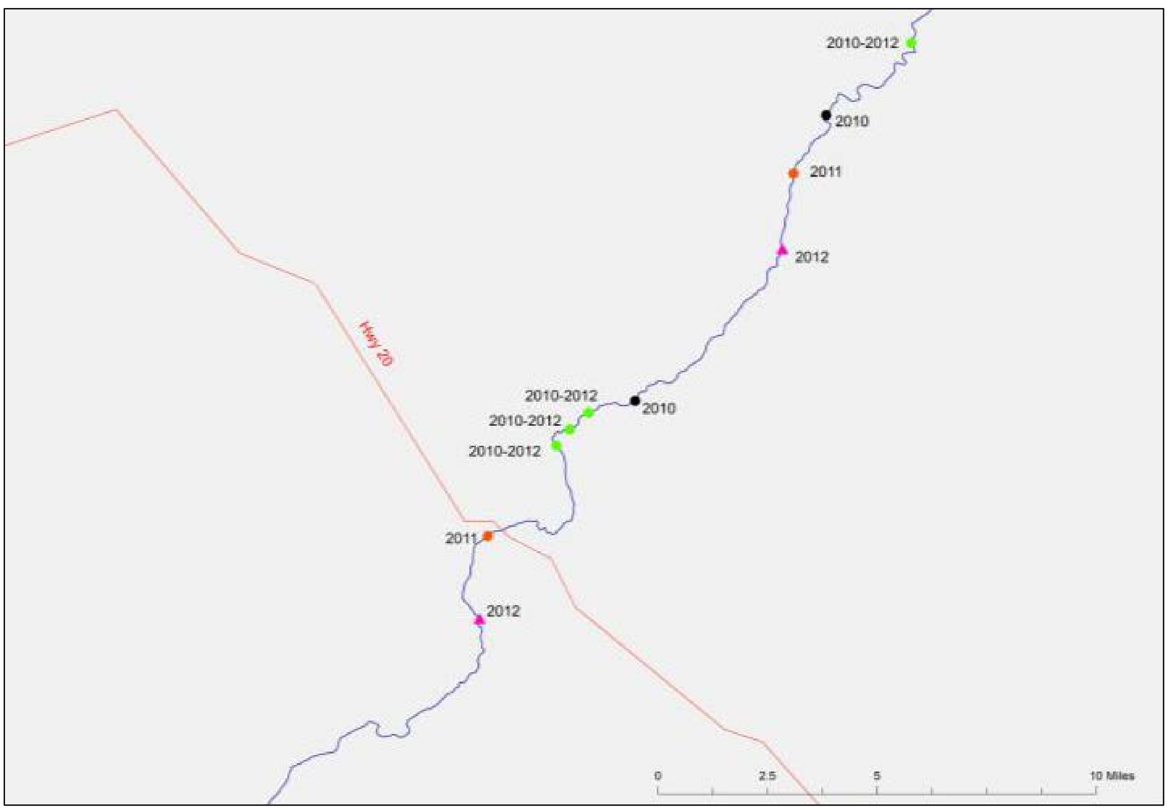

Figure 3.
Redband redds were counted in eight reaches in 2012: two designated index reaches (green circles: reach 1, rkm 2; reach 8, rkm 27); two randomly selected reaches (pink triangles: Rimrock Ranch, rkm 14; upstream of Sisters, rkm 38); and in four original reaches (green circles: reach 2, Alder Springs Creek; reach 3, Lewis Woodpecker Creek; reach 6, rkm 25; and reach 7, rkm 26) surveyed from 2006-2012. Reproduced with permission from Quesada *et al* 2013.

Results

Species Composition

As in previous years, the majority of fish captured in Whychus Creek in 2012 were *O. mykiss* including both resident redband and released steelhead. Other species captured included Chinook salmon parr, brown trout, sculpin, and longnose dace.

O. mykiss Population Estimates

Whychus Creek *O. mykiss* population estimates continue to vary widely between years, with no statistically significant increasing or decreasing trend detectable over six years of sampling (Table 2). Estimated density for each study reach has fluctuated between years with no consistent pattern between reaches (Figure 4). Size distribution of *O. mykiss* from 2007-2012, since steelhead reintroduction, 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 2007-2012. 2006 data 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
1 (Alder Springs)	48 (± 28)	24 (± 24)	12 (± 4)	11 (± 4)	24 (± 5)	7 (± 2)
2 (Road 6360)	25 (± 10)	9 (± 3)	24 (± 9)	13 (± 4)	15 (± 3)	15 (± 3)
3 (Camp Polk)	60 (± 13)	52 (± 21)	57 (± 15)	27 (± 9)	16 (± 3)	-
4 (Sisters)	20 (± 10)	5 (± 2)	23 (± 14)	18 (± 6)	10 (± 5)	28 (± 8)
5 (Wolftree)	-	-	21 (±7)	106 (± 29)	42 (± 9)	32 (± 9)
USFS site at TSID	-	-	2.4 (1.5-4.0)	-	-	-

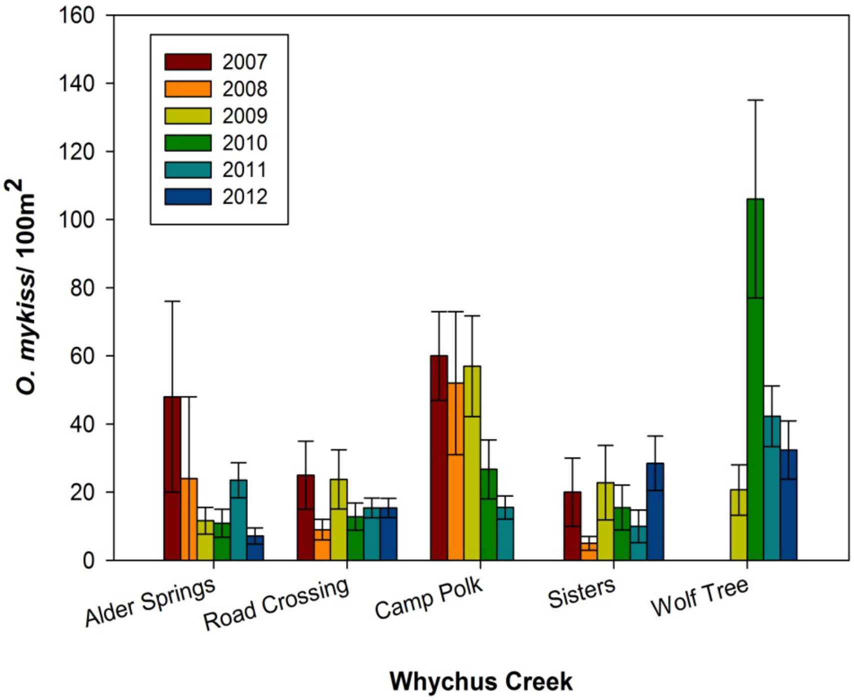

Figure 4.
O. mykiss densities for five Whychus Creek sites from 2007 to 2012. Reproduced with permission from Quesada *et al* 2013.

Spring Chinook Juvenile Density

Mark-recapture surveys resulted in 2012 density estimates of 0-10 juvenile Chinook/100m^2. These numbers are down from 2009-2011 estimates.

Table 3. Spring Chinook densities in Whychus Creek in 2009-2011 estimated from mark-recapture and snorkel surveys.

	Spring Chinook Density (Fish/100m^2)						
	2009		2010		2011		2012
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
2 (Road 6360)	4.7 ± 2.3	8	1 ± 0.5	2	2 ± 1	0	1.2 ± 0.5
3 (Camp Polk)	17.4 ± 14.7	--	0	--	0	--	--
4 (Sisters)	2 ± 0.8	4	0	4	0	--	10.1 ± 7.9
5 (Wolftree)	8.3 ± 6.1	0	25 ± 16	4	15 ± 19	1	0.8 ± 0.0

Smolt production

Despite operating throughout the migration season, PGE was not able to capture enough fish in 2012 to generate a population estimate for the Upper Deschutes. The confluence of the Deschutes River and Lake Billy Chinook is problematic for trapping effectively and safely, and the native fish monitoring team was unable to place a Merwin trap in the thalweg at this location until the reservoir reached full pool (M. Hill personal communication, 2013).

O. mykiss Redd Surveys

Redd counts conducted in Whychus Creek during the 2012 spawning season were compromised by high turbidity and consequent low visibility downstream of Camp Polk, attributed to the stream channel restoration project completed in February the same year, and at Alder Springs, attributed to a 2011 wildfire. A total of eight redds were detected in the Camp Polk area (Figure 5). Nate Dachtler with Deschutes National Forest additionally observed spawning in the reconstructed meadow channel at Camp Polk. Although four redd counts were conducted at Rimrock Ranch, surveyors were unable to see the substrate clearly, and no redds were detected at this site. Seven redds were detected in the Alder Springs area in 2012, as compared to 24 in 2011.

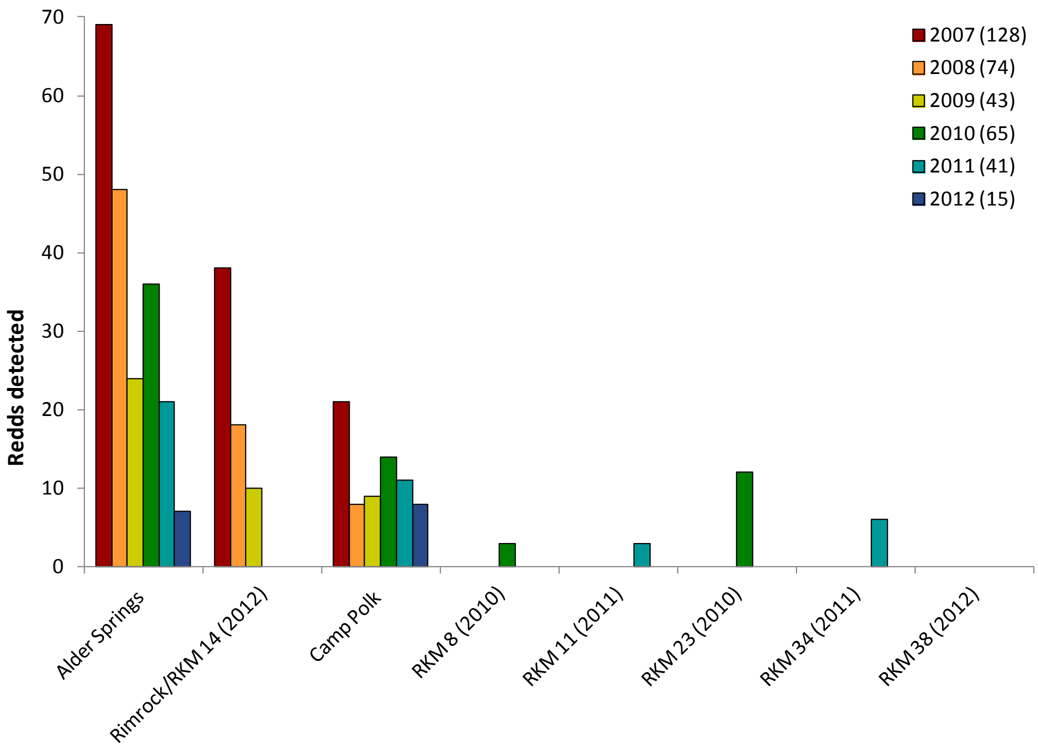

Figure 5.
Redband redds detected by site and year in Whychus Creek. Totals for each year are shown in the legend.

Discussion

Population Estimates

Despite annual releases averaging 268,530 fry and 4,730 smolts, comparable to numbers released in McKay and Ochoco Creeks, juvenile *O. mykis*s density remains low in Whychus, and far lower than densities estimated for the two Crooked River tributaries (Quesada *et al* 2013). While overall juvenile *O. mykiss* densities in Whychus remained low in 2012 and densities decreased from previous years at two sites (Alder Springs and Wolf Tree), densities at two other sites (6360 Road Crossing and Sisters) were consistent with or higher than those observed in 2010 and 2011.

One possible explanation for low *O. mykiss* density is high juvenile mortality or fry flushing out of the creek as a result of the combined effects of high flow events and low availability of off-channel habitat and habitat complexity to provide refuge during such events. Despite the continued low estimates for O. mykiss density, estimates remain higher than 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). This overall increase over pre-introduction levels is consistent with the continued annual releases of steelhead fry and smolts.

Poor spawning and emergence conditions in 32% of river miles surveyed for habitat quality between 2008 and 2011 (10.5 of 32.3 miles surveyed), reflecting fine sediment conditions in excess of the range of *O. mykiss* spawning criteria and low gravel and cobble percentages (Mork 2013a), could contribute to low abundance of juvenile resident redband spawned and emerged in Whychus Creek which in turn could account in part for the low juvenile densities observed in Whychus, but any such contribution is likely minimal given the low density of redband trout prior to steelhead reintroduction. Macroinvertebrate data from 2005 to 2012 (Mazzacano 2012) indicate impaired, if improving, conditions at most sites, providing some evidence for compromised stream conditions that may support lower numbers of juvenile O. mykiss than otherwise anticipated.

The inability to differentiate between juvenile redband and steelhead continues to limit conclusions about the relative abundance of the two life histories and interactions between the two populations. In 2010 PGE collected genetic samples from a subset of *O. mykiss* captured during electrofishing that may be used in the future to differentiate between juvenile steelhead and redband. PGE did not retain fin clips in 2012 nor in 2011 from juvenile *O. mykiss* captured in Whychus Creek (contrary to what we reported in the 2011 Native Fish Monitoring in Whychus Creek report)(Mork 2013b). As noted previously, PGE is scheduled to conduct analyses to differentiate between juvenile redband and steelhead five and ten years (in 2017 and 2022, respectively) after returning steelhead were first passed upstream of the dams in 2012. The ability to differentiate between juvenile steelhead and redband will allow researchers to better understand the status and trends of resident redband and reintroduced steelhead populations, potential effects of competition between the two life histories, and spawning interactions.

Spring chinook densities increased substantially at the Sisters site in 2012 over densities recorded in previous years (Table 3), while densities at Wolftree plummeted in keeping with the overall decrease in Chinook density since fry and smolts were first released in 2009. As no Chinook are known to have naturally spawned in Whychus Creek as of 2012, all Chinook observed in Whychus were likely released as fry or smolts. Although spring Chinook rearing and migration in the Upper Deschutes subbasin is not well documented, spring Chinook in the Lower Deschutes outmigrate in the spring at age 0+ or 1+. Thus, density estimates for juvenile spring Chinook likely represent almost exclusively fish outplanted the same year. Low Chinook numbers may accordingly be a function of sampling timing, i.e. later in the season than most Chinook would be expected to have outmigrated.

Smolt production estimates

As in previous years, high water impacted trap operations in 2012, preventing PGE from estimating smolt production for the Deschutes. PGE will continue to refine smolt trapping methods to obtain a smolt production estimate for the Deschutes.

O. mykiss Redd Counts

Redd surveys were highly ineffective in 2012 due to exceptionally high turbidity and low visibility. Only twelve redds were detected over eight sites, compared to 41 redds detected in 2011 and 65 detected in 2010 using the same sampling design. Redd survey data from 2007-2011 shows redband spawning in

Whychus Creek occurring from March through July, peaking in May in three out of five years surveyed (2007, 2008, 2010), in April in 2009, and in June in 2011. The high number of redds detected in 2007 corresponds to the only sampling year during which high and turbid flows did not impede surveys, suggesting that the 128 redds detected that year may be most representative of redband spawning numbers in Whychus Creek. The greatest proportion of redds have consistently been observed in the Alder Springs Area, from 51% in 2011 to 65% in 2007. Rimrock Ranch, Reach 5, accounted for approximately a quarter of redds detected (23-29%) during the years it was surveyed, from 2007 to 2009. Together, Alder Springs and Rimrock Ranch accounted for 80-90% of all redds observed from 2007 to 2009, suggesting these areas were the primary spawning grounds of reaches sampled in Whychus Creek for redband in these years. The Camp Polk area, from rkm 25 to rkm 27, accounted for approximately 20% of all redds detected in both 2009 and 2010, increasing to 27% in 2011. Only one redd was observed between 2007 and 2009 in reaches 9 and 10 upstream of Sisters; low numbers in these reaches were attributed to upstream and downstream barriers inhibiting migration of spawning fish. However, in 2011 six redds, or 15% of the 2011 total, were counted in a nearby reach (rkm 34).

Information on numbers and distribution of spawning fish have already helped to document redband reproduction trends and spawning habitat use in Whychus Creek, and will provide a baseline for O. mykiss spawning activity and interactions when adult steelhead return in greater numbers. Redband redd measurements collected prior to the return of spawning steelhead to Whychus will provide a basis for differentiating between redband and steelhead redds.

Conclusions

Although fish population data from 2007 to 2012 reveal no clear O. mykiss population trends in Whychus Creek, the decline in *O. mykiss* indicated by 2007-2011 data stabilized in 2012, with an average 2012 density estimate equivalent to the 2011 average. The number of redband redds detected has fallen in every year except 2010. Average spring Chinook density estimates and snorkel counts have fallen every year since Chinook were first reintroduced in 2009.

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

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

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

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