Upper Deschutes Watershed Council
Technical 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
PBACI	Paired Before After Control Impact
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 the 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 2011 Whychus Creek Monitoring Report assesses indicators in one of these categories.

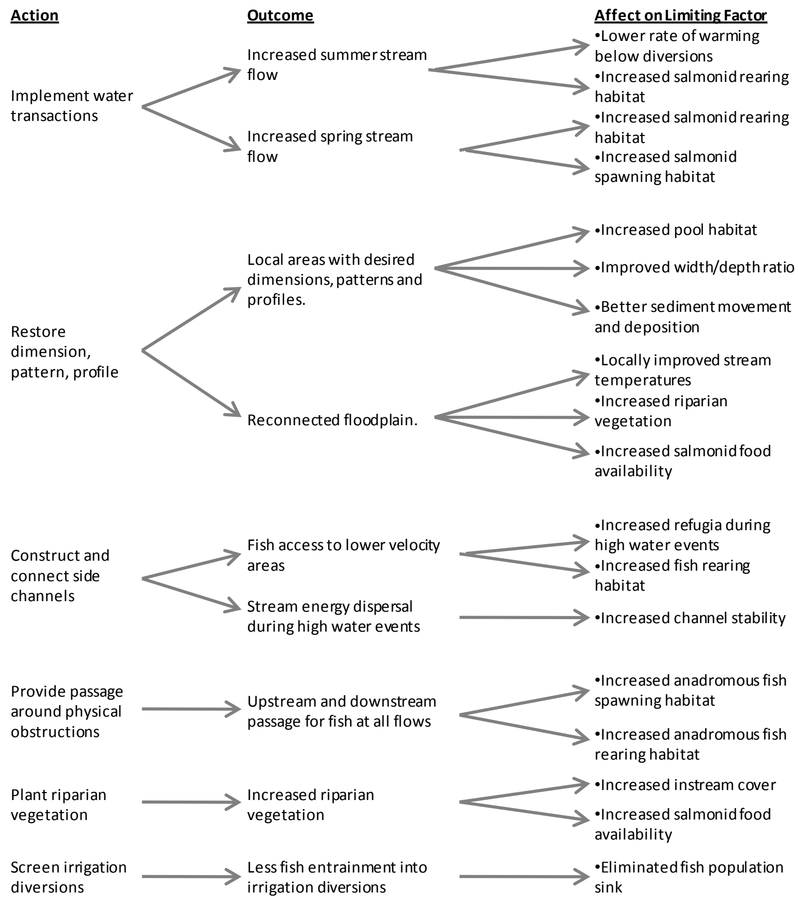

Figure 1.
This conceptual model illustrates the expected influences on each life stage of resident and anadromous salmonids in Whychus Creek. The UDWC expects that the ongoing restoration actions will affect the limiting factors identified in the conceptual model.

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% (UDWC 2000). Tributaries to Whychus Creek include Snow Creek, Pole Creek, and Indian Ford Creek.

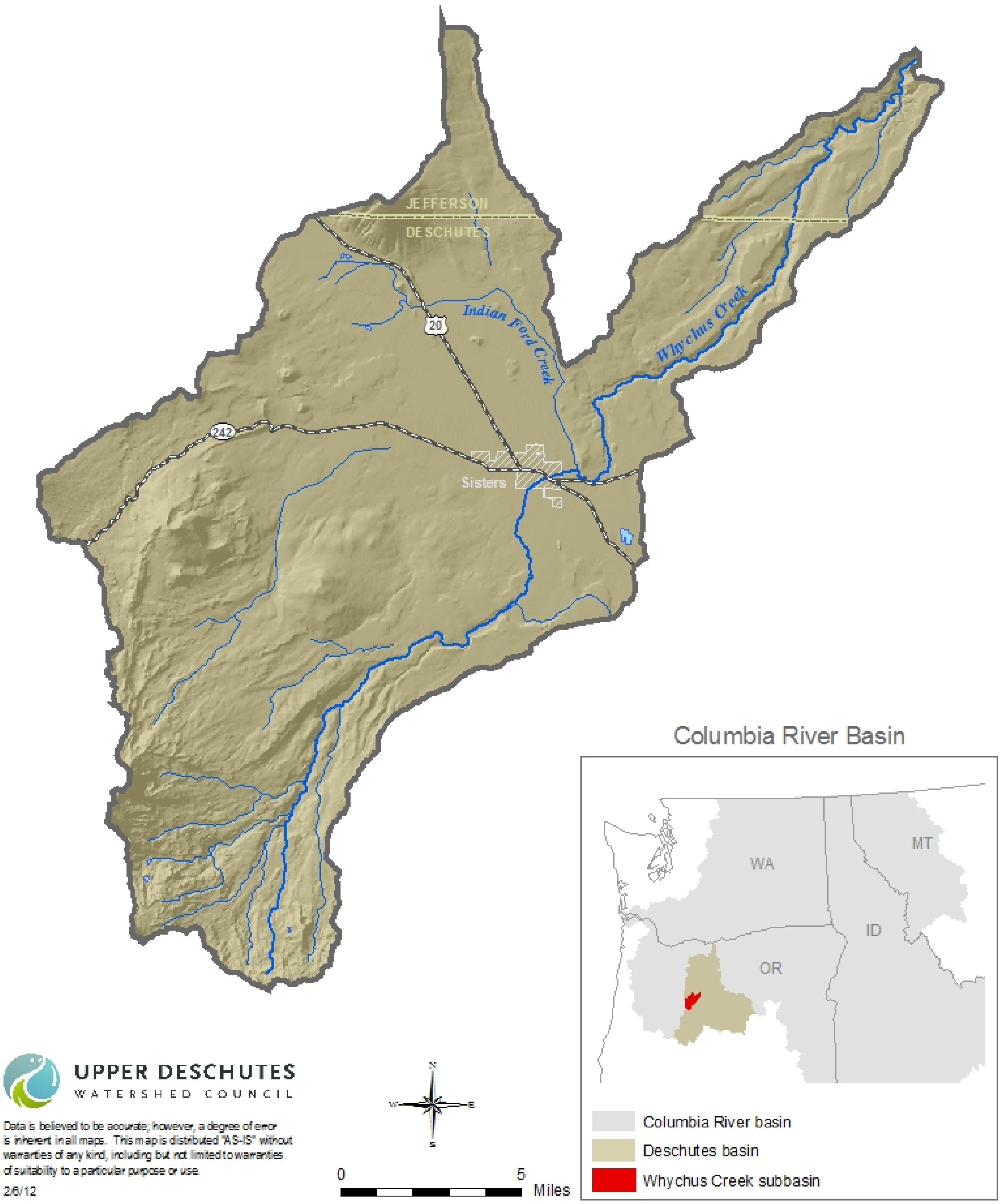

Figure 2.
Whychus Creek extends from the Cascade Range to the Deschutes River. The creek's watershed encompasses approximately 162,000 acres of Deschutes and Jefferson Counties in central Oregon

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 the 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 2000-2011. It focuses on metrics representing low flow conditions in the creek. Mork (2013a) answers questions related to stream temperature in Whychus Creek. It draws from thirteen years of data to examine water quality in relation to state standards and to stream flow restoration. Restored stream flow has likely affected metrics in each of these reports.

A third report updates the habitat quality ratings and analysis presented in the 2009 baseline monitoring report (Mork 2013b). Although restoration work completed during the scope of the analysis is not expected to directly affect the habitat quality metrics evaluated, this report refines baseline information and establishes a foundation for future habitat quality analyses.

Two reports quantify habitat improvements resulting from restoration projects completed subsequent to baseline analyses. Mork (2013c) 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 (2013d) 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.

Two additional reports update the status of biological conditions in the creek. Mazzacano (2013) examines four years of macroinvertebrate data to identify trends in macroinvertebrate community composition before and following extensive streamflow restoration. Mork (2013e) summarizes PGE's 2011 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.

These seven 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 action at moving the creek toward desired conditions. Restoration partners expect to draw from these reports to continually improve restoration implementation and monitoring in the 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. 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 58 cfs in 2009. 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. These results highlight the need to understand whether low flows during these periods limit ecosystem function and, if so, to focus on restoration efforts during these periods. As restoration continues to increase flows in Whychus Creek, restoration partners should continue to evaluate both early and late season flow as well as extreme low flows to fully describe restoration outcomes.

Introduction

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

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

Monitoring the status and trends of stream flow in Whychus Creek will illuminate whether the stream is moving towards or away from desired conditions.

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

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

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

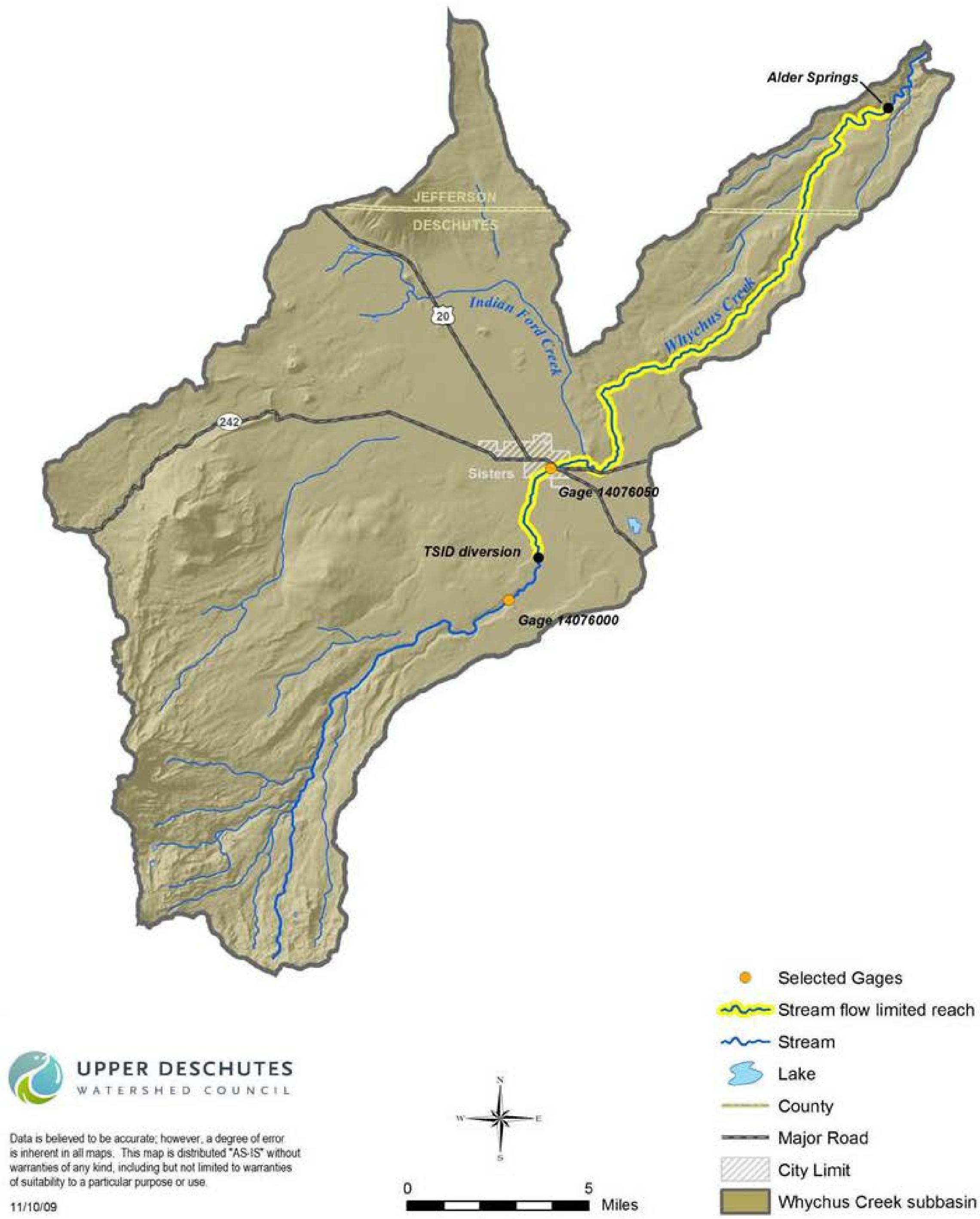

Figure 1.
Stream flow limits stream function in Whychus Creek downstream from the Three Sisters Irrigation District Diversion. Spring inputs near the mouth of Whychus Creek increase stream flow and improve conditions in the creek.

Metric	Appears In
30 day minimum	Gao *et al* 2009, Richter *et al* 1996
May median flow	Gao *et al* 2009, Richter *et al* 1996
August median flow	Richter et al1996

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

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

Minimum 30 Day

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

May Median

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

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

August Median

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

Methods

Data Collection

The Oregon Water Resources Department (OWRD) maintains several gages along Whychus Creek. They operate gage 14076050 at the City of Sisters, downstream from major irrigation diversions along the creek (Figure 1). OWRD began operating this gage in 2000 and has continued operating it through the publication of this report in 2011.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 2011. 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 2011 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 2009a). 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, 2011 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 2011. 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 2011.

May Median

The DRC used spreadsheet software to determine the median daily average flow during the month of May for years 2001 through 2011. The DRC only had partial data for 2000 and did not include that data in this analysis.

<u>August Median</u>

The DRC used spreadsheet software to determine the median daily average flow during the month of August for years 2000 through 2011. 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

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 58.0 cfs in 2009. 2006 exhibited the greatest intra-annual variation in May flow, with a 20^{th} percentile value of 22.0 cfs and an 80^{th} percentile value of 122.0 cfs.

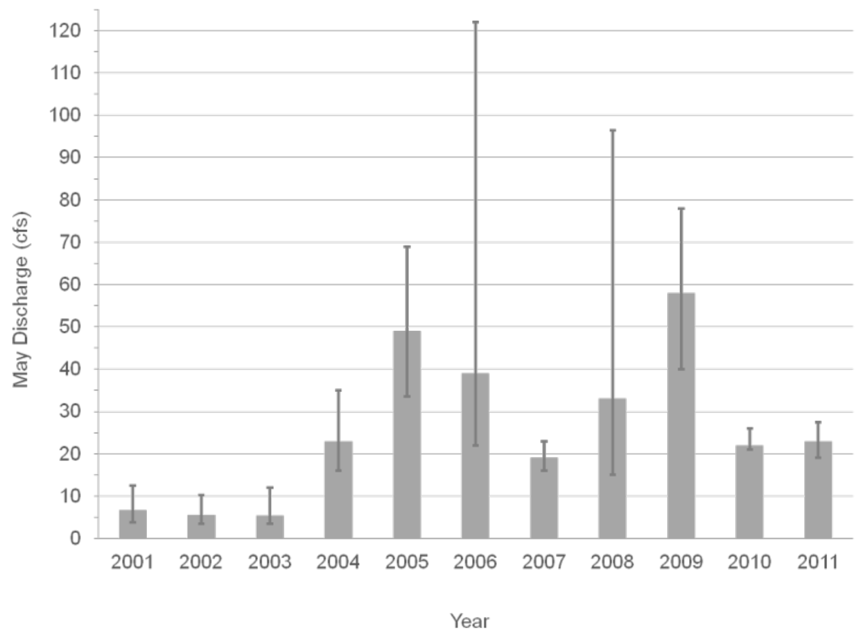

Figure 2.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of May provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of May at this location.

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.

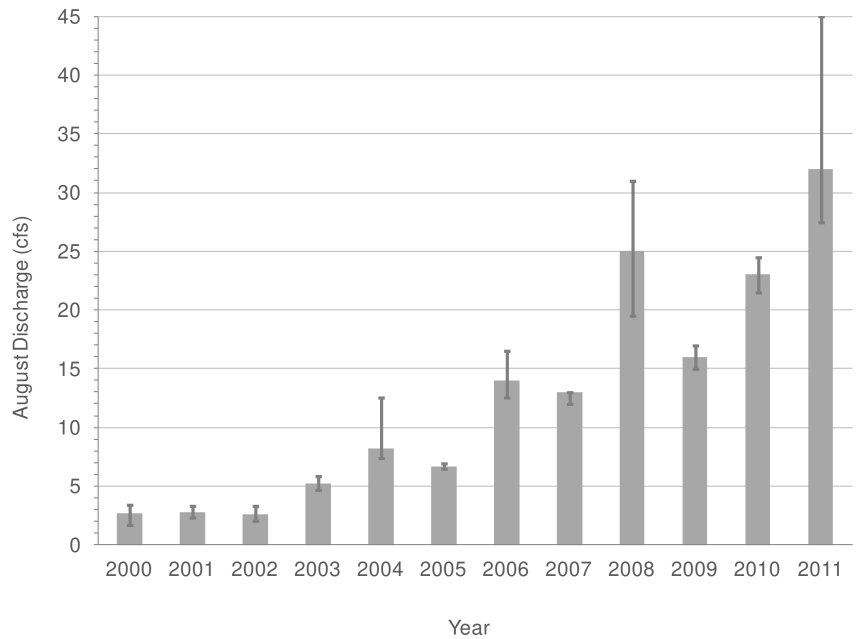

Figure 3.
The median of the average daily discharge of Whychus Creek at the Oregon Water Resources Department's gage number 14076050 during the month of August provides one indicator of low flow magnitude. Error bars represent the 20th and 80th percentile discharges during the month of August at this 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 2011. 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 1996, Figure 2). It never met Oregon's March, April, and May instream water right of 50 cfs for Whychus Creek downstream from Indian Ford Creek (OWRD 1996, Figure 3).

Restoration partners have focused on late summer stream flow as a metric for restoration effectiveness. Late spring/early summer stream flow may also be important for stream function. As noted earlier, redband trout spawning centers on the month of May (ODFW 2005). Consistently low stream flow 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.

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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 2011 at eleven sites representing the diversity of flow conditions in Whychus Creek. This report incorporates 2011 data to evaluate the current 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 target flows projected to produce temperatures that meet state standards. Temperatures exceeded state rearing and migration standards at three monitoring sites in 2011 for a total of 19 days, down from five sites and 49 days in 2010 and six sites and 47 days in 2009. Temperatures never exceeded the lethal threshold for salmon and steelhead in 2011, for the second year in a row. Before-After-Control-Impact (BACI) analysis of differences in stream temperature between years indicated that temperatures between the Three Sisters Irrigation Diversion (TSID) and Alder Springs cooled from 2010 to 2011 in response to streamflow restoration, with temperatures downstream of Alder Springs warming over the same interval, supporting trends observed from 2002-2006 and 2002-2010. Opposite trends observed from 2006 to 2010 (warmer temperatures downstream of TSID and cooler temperatures downstream of Alder Springs) were attributed to a reduction in actual flow during this period. Regression of 1995-2011 temperature and flow data identify 66 cfs as the minimum flow necessary to meet the 18° temperature standard at FS Road 6360, corroborating previous results. These results provide a scientific basis for adaptive management and restoration planning in Whychus Creek.

Introduction

Restoration partners have identified the Whychus Creek watershed as a priority watershed for conservation and restoration within the upper Deschutes Basin (NWPPC 2004, UDWC 2006). Diversion of almost 90% of average summer flows and historic channelization of nearly 50% of the creek length create conditions that contribute to elevated stream temperatures and may compromise other water quality parameters. In 1998, 2002, and 2004, 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). In May 2011 ODEQ submitted their 2010 Water Quality Assessment Integrated Report to the EPA (ODEQ 2011). The 2010 assessment for Whychus Creek is consistent with ODEQ's 1998, 2002 and 2004 findings.

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 temperatures naturally increase as water flows downstream, and temperatures 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 13˚C state temperature standard for salmon and steelhead spawning. The state temperature standard for salmon and trout spawning is 13˚C. Neither the 2004 nor the 2010 303(d) lists applied this criterion to Whychus Creek because anadromous fish were not present 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. 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 salmon and trout egg development, egg hatching, and embryo development, with consequent effects on aquatic insect populations. 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. 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 cool, 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, documented preliminary changes in temperature observed with streamflow restoration, and developed a regression model describing the relationship between temperature and flow to identify streamflow restoration targets. The subsequent version of this report incorporated 2009 and 2010 data to update the 2006-2008 analyses (Mork 2012). We present revised analyses including 2000-2011 temperature and flow data to evaluate the current 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 target flows 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	12.8° 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

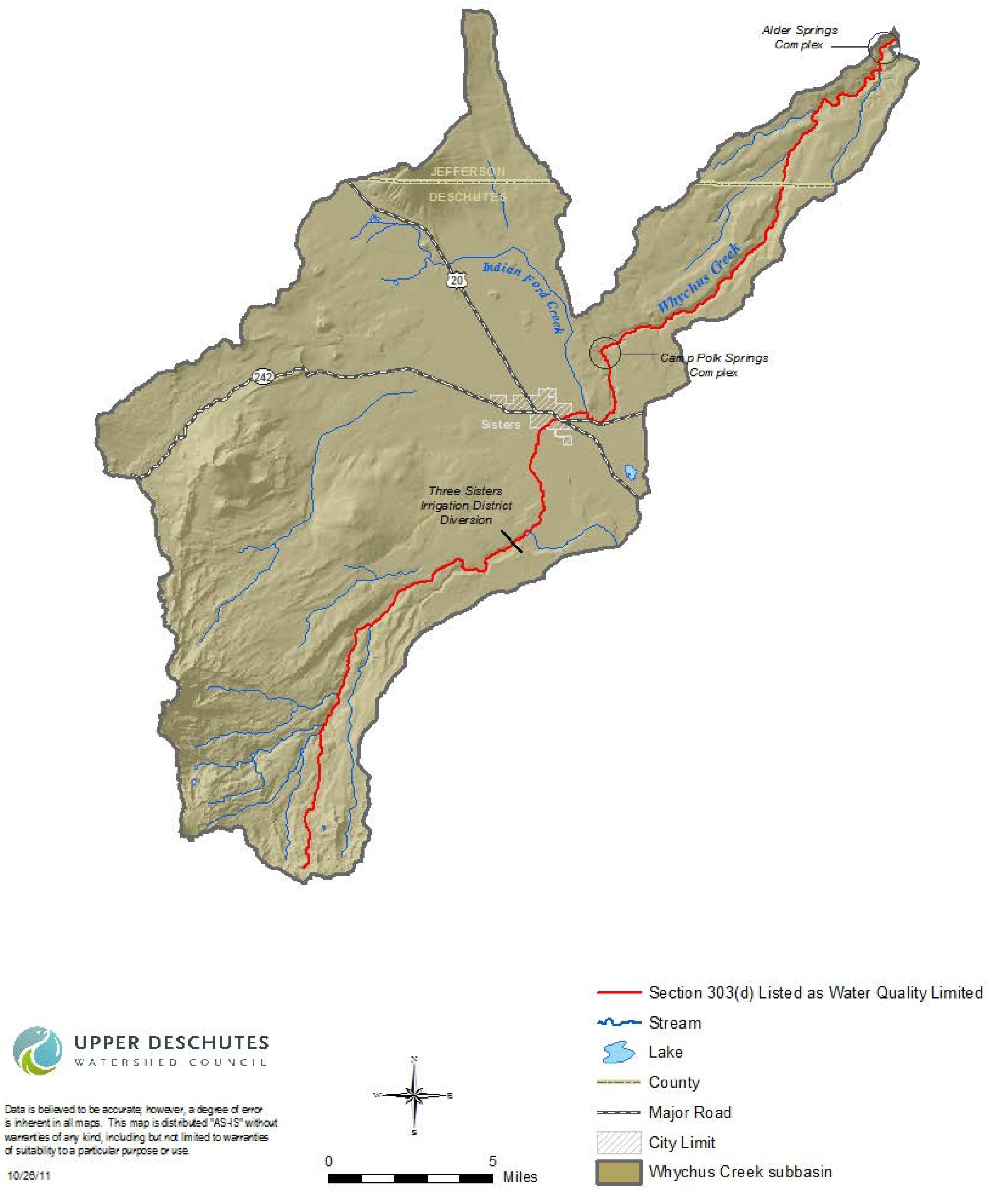


Figure 1.
Whychus Creek is listed as Water Quality Limited from river mile (RM) 0.0 to RM 40.3 under ODEQ's 2002, 2004, and 2010 303(d) lists. (ODEQ 2011).

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 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 2011). This gage is located downstream from the Three Sister's Irrigation District diversion and other major irrigation diversions. We used data collected at this gage from 2000 to 2011 in this report, including some data considered by OWRD to be provisional and subject to change.

Temperature status

We evaluated 2011 seven day moving average maximum daily temperatures (7DMAX) in relation to the 18˚C state temperature standard for salmonid rearing and migration and the 13˚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 the 18˚C standard, we compared temperatures to flows recorded at Sister's City Park in relation to ODFW water rights and DRC streamflow restoration targets. We also compared the status of Whychus Creek temperatures in 2011 to 2000-2010 results.

We calculated the average rate of temperature change for Whychus Creek on the hottest water day of 2011 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.

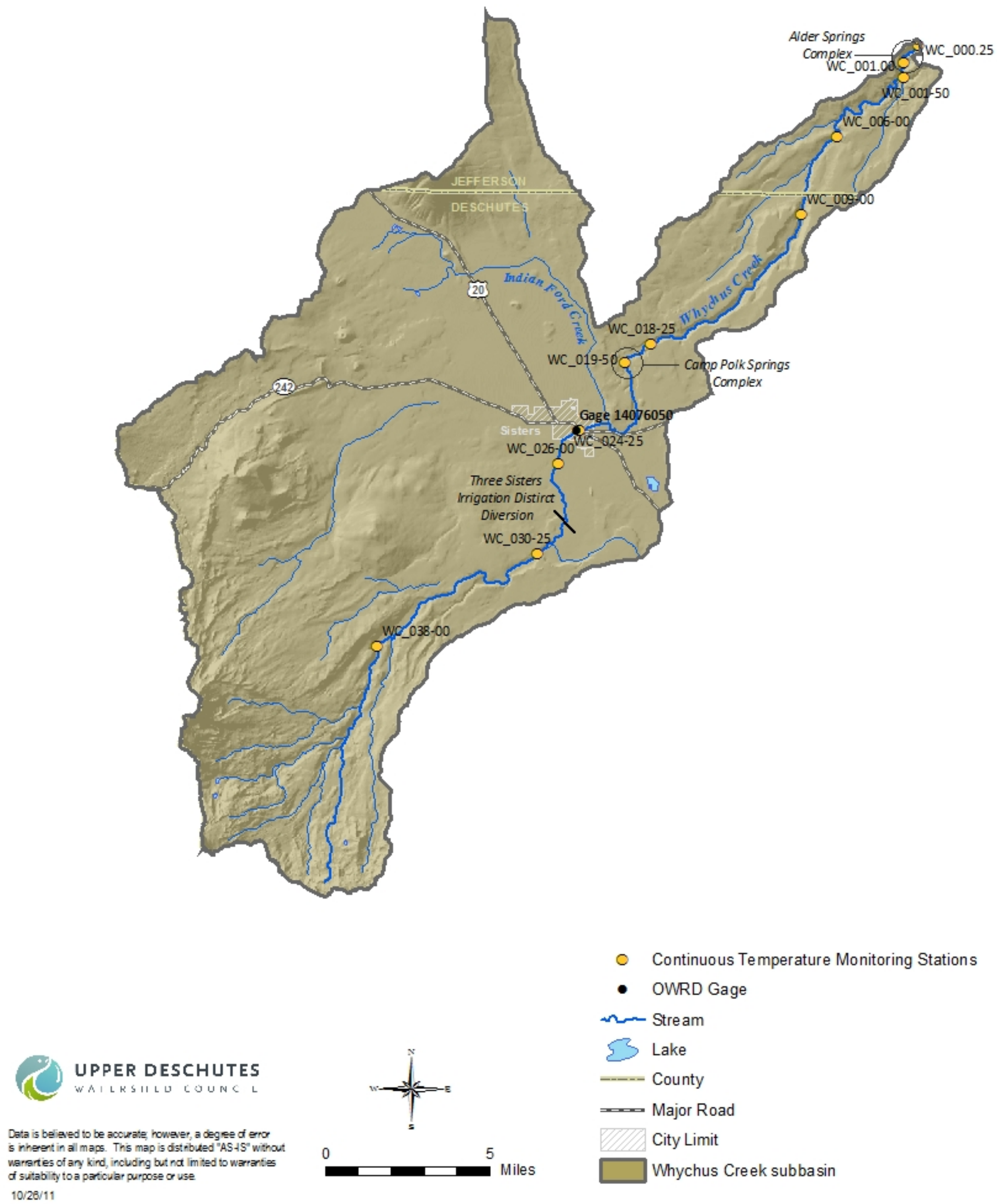


Figure 2.
Continuous temperature monitoring stations monitored in 2009 and 2010 and OWRD Gage 14076050 on Whychus Creek.

Streamflow restoration effectiveness

Five locations on Whychus Creek where UDWC has monitored temperature continuously since 2002 have served as reach boundaries in analyses quantifying the local and longitudinal effects of restored flows on stream temperature from 2002 to 2010 (Figure 2). These stations define the boundaries of a reference reach and three restoration reaches. The reference reach (WC 038.00 – WC 030.25) represents natural flow conditions upstream of all irrigation diversions. Restoration reach 1 (WC 030.25 – WC 024.25) shows local impacts of streamflow restoration immediately downstream of the most significant irrigation diversion on the creek and the site of significant streamflow restoration; restoration reach 2 (WC 030.25 – WC 006.00) demonstrates longitudinal effects of restored flows 18 miles downstream of the same site; and restoration reach 3 (WC 030.25 – WC 000.25) illustrates the longitudinal effects of streamflow restoration after significant coldwater inputs from Alder Springs. We used these four reaches for 2002-2006, 2006-2010, 2002-2010, and 2010-2011 analyses.

In 2011 we included data from a sixth location (WC 001.50) to isolate the temperature response to flow restoration above and below Alder Springs from 2010 to 2011. We used this station to create two additional restoration reaches. Restoration reach 4 (WC 030.25 – WC 001.50) demonstrates the longitudinal effects of restored flows 22.5 miles downstream of the site of significant streamflow restoration, instead of the 18 miles downstream measured by restoration reach 2. Restoration reach 5 (WC 001.5 – WC 000.25) quantifies the effects of streamflow restoration over 1.25 miles where Alder Springs flows contributes 95 cfs of cold groundwater to the volume of Whychus Creek, removing the effect of streamflow restoration on temperatures in the 28.75 miles between WC 030.25 and WC 001.5 and upstream of the Alder Springs coldwater inputs.

To control for natural variability in streamflow, climate (e.g. precipitation, solar insolation, air temperature, etc.) and other environmental factors that influence inter-annual differences in temperature we used a paired before-after control-impact (BACI) design that compares pre- (Before) and post- (After) restoration changes between years within a reference (Control) reach to changes between years within a restoration (Impact) reach (Smith 2002). By accounting for inter-annual environmental variability this analysis allows differences in temperature observed between reference and restoration reaches to be attributed to the effects of streamflow restoration. We compared July 2011 temperature data to data from July 2010 to quantify the temperature response in restoration reaches to increased flows resulting from streamflow restoration from 2010 to 2011. We restricted data included in the analysis to one month of the year to reduce the effect of inter-annual seasonal variation in the analysis (Helsel and Hirsch 2002) and selected July as the month during which the hottest water day has historically occurred most frequently (UDWC 2003; UDWC 2008c). We calculated values for analysis from daily median temperatures, a statistic which reflects small changes in temperature more precisely than the daily mean or daily maximum temperature, and at a finer time scale than the seven day moving average maximum temperature (7DMAX).

To calculate BACI differences we subtracted 2011 from 2010 temperatures for each station (e.g. WC 038.00 $_{2010}$ - WC 038.00 $_{2011}$) then subtracted the downstream from the upstream difference for each reach (e.g. Δ WC 038.00 $_{2010\text{-}2011}$ – Δ WC 030.25 $_{2010\text{-}2011}$) to quantify the longitudinal change in water temperature. We compared the mean BACI difference of changes between years in the reference reach to the same difference in the four restoration reaches to evaluate how restoration reaches changed between data years relative to the reference reach.

Analyses were conducted using R open source statistical software (R Core Development Team 2007).We used normal plots and a Shapiro-Wilk test to establish normal distribution of data. Where data were normally distributed we used a t-test to to identify 1) whether temperature changes observed between years in restoration reaches were significantly different than changes observed between years in reference reaches, and 2) in which direction these changes occurred (warming or cooling) relative to the reference reach (Helsel & Hirsch, 1991) . For reaches where data were non-normal we used an exact permutation test (Hothorn and Hornik 2006) to compare the restoration reach and reference reach means. An exact permutation test for paired samples compares the observed statistic, the difference of means from two experimental groups, to the expected statistic under a permutation distribution created by randomly resampling from all possible permutations of the data from treatment and control groups. Here the observed statistic is the difference of the restoration and reference reach means. For each restoration reach- reference reach pair and for each combination of data years we evaluated the following four hypotheses:

1) H_0: There is no difference between the mean for the restoration reach and the mean for the reference reach.

2) H_1: The mean for the restoration reach and the mean for the reference reach are statistically different.

3) H_2: The mean for the restoration reach is significantly less than the mean for the reference reach; the restoration reach has cooled relative to the reference reach

4) H_3: The mean for the restoration reach is significantly greater than the mean for the reference reach; the restoration reach has warmed relative to the reference reach.

We compared 2010-2011 results to relative changes in temperature observed from 2002-2006, 2006-2010, and 2001-2010.

Target Streamflow

We included 2011 data with our 2000-2010 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. As in our restoration effectiveness analysis, we restricted data included in the regression to one month of the year to reduce the effect of inter-annual seasonal variation (Helsel and Hirsch 2002) and selected July as the month during which the hottest water day has historically occurred most frequently (UDWC 2003; UDWC 2008a). We used July 7DMAX temperature data for each year included in the analysis from two monitoring stations, WC 024.25 and WC 006.00, 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 diversion of all but protected flows; 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. Daily streamflow data for all July days from 2000-2011 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 2011).

To describe the relationship between flow and temperature at the two locations we performed a regression of temperature and flow data. The resulting equations accurately represent the relationship between flow and temperature only for the specific 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 data and assigned associated temperatures from all July days to each flow value, excluding flows with one or no corresponding temperature record ($n \leq 1$), to calculate the mean of all 2000-2011 July 7DMAX temperatures observed at each flow level. We plotted flows versus mean temperature and fitted a regression trendline that best described the data by adding polynomial terms to the corresponding regression equation. We evaluated S and R^2 values to assess the fit of the regression model to the temperature-flow data. S is the standard error and represents the standard distance (°C) that mean

7DMAX temperature values fall from the regression line. A better fit between the regression line and the data results in a lower S value. 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 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 2000-2011 temperature-flow regressions and predicted temperatures at given flows for each site to 2000-2010 regression models and 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 2000-2011 temperature calculations for these flows.

Results

Temperature status

Although seven-day moving average maximum (7DMAX) temperatures continued to exceed the 18˚C state standard for steelhead and salmon rearing and migration at some sites in 2011, July mean 7DMAX temperatures at Sisters City Park (WC 024.25) chart a decreasing trend since 2002. July 7DMAX temperatures at this site, three miles downstream of the most significant irrigation diversion on the creek, remained below the 18˚C standard in 2011 for the third year in a row, with only one year on record exceeding the state standard since 2006 (Figure 3). Temperatures remained below the 18˚C standard throughout July 2011 at all sites monitored, exceeding the state temperature standard only in August 2011, at three downstream sites between RM 8.75 and RM 1.5, down from five sites in 2010 and six sites in 2009 (Figure 4). 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 26 to 39 cfs for 19 days in 2011, down from 49 days in 2010 and 47 days in 2009. Temperatures exceeded the 18˚C standard for a total of 20 days, the most days at any site in 2011, at Rimrock Ranch (WC 008.75). Lethal temperatures were not recorded at any site on Whychus Creek in 2011, for the second year in a row.

Continuous temperature data were available for the 2011 spawning season from April 30 to May 15 for two downstream sites (WC 000.25 and WC 001.50) and from May 2 to May 15 for the remaining eight sites (Figure 4). Temperatures recorded for these dates exceeded the 13˚C spawning habitat requirement and potential state standard for 29% to 93% of spawning season days for which data were available at five sites between RM 18.25 and RM 1.5 in 2011, down from six sites between RM 19.5 and RM 1.5 in 2010. Temperatures exceeded the 13˚C habitat requirement for 13 days, the most days at any site in 2011, at Rimrock Ranch (WC 008.75).

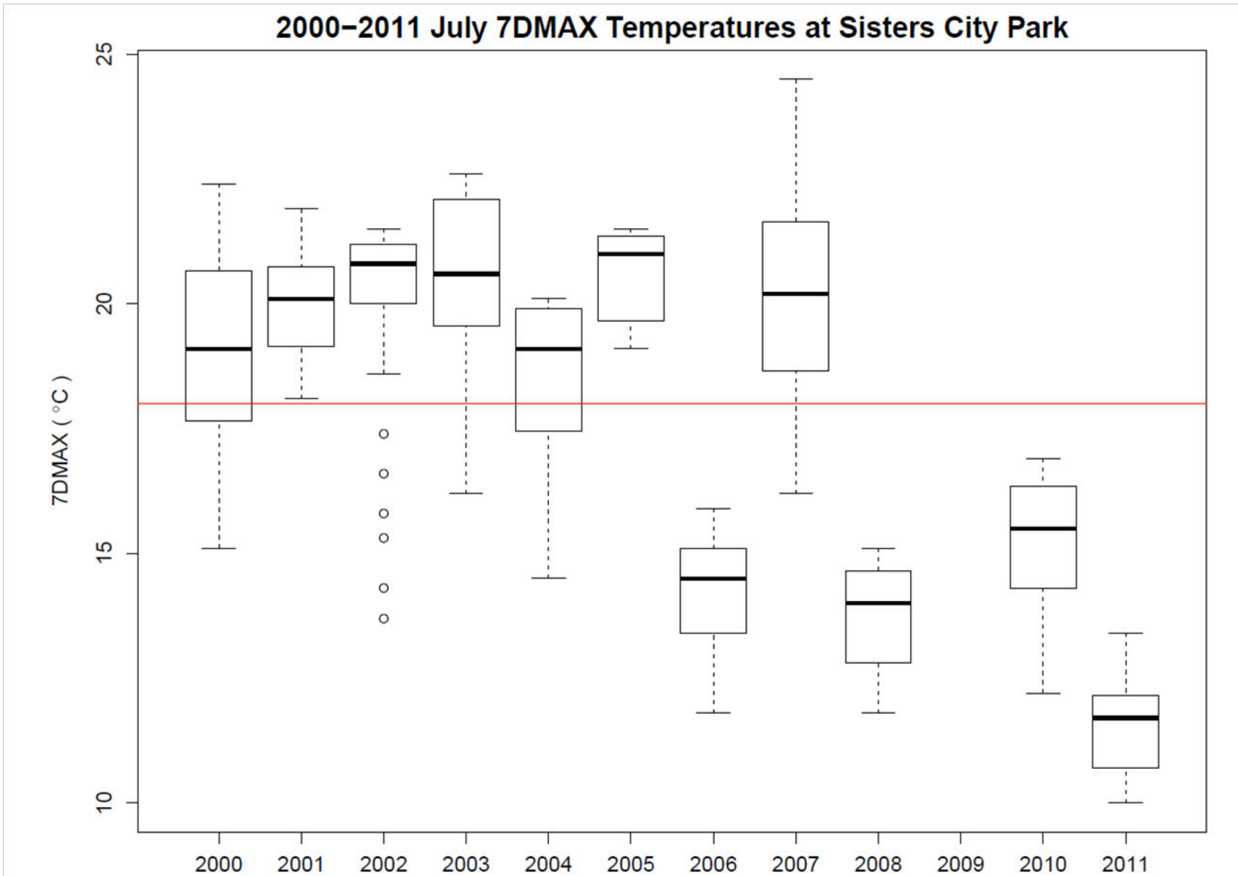


Figure 3.
July mean 7DMAX temperatures at Sisters City Park (WC 024.25) chart a decreasing trend since 2002. Temperatures at this site have exceeded the 18˚C state standard in only one year for which data are available since 2006. No data are available for this site for 2009.

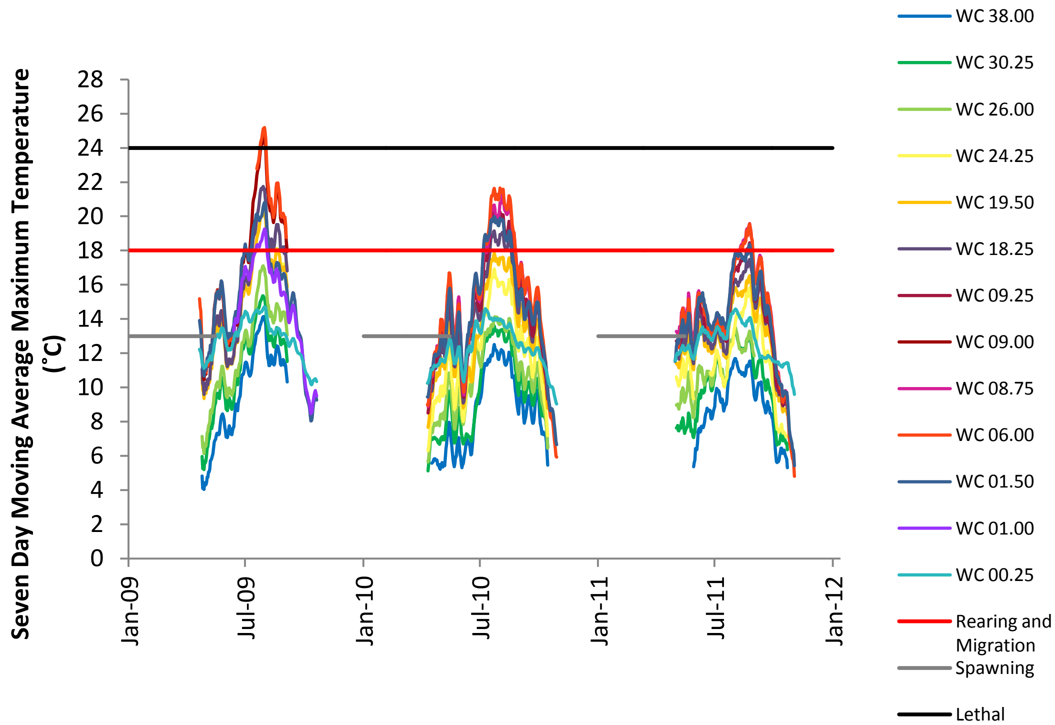


Figure 4.
Temperatures exceeded the 18˚C state standard for salmon and trout rearing and migration at three monitoring locations in 2011, down from five locations in 2010 and six locations in 2009. Temperatures exceeded the January 1- May 15 13˚C spawning standard at five sites in 2009 and 2011 and at six sites in 2010. Temperatures exceeded the 24˚C lethal threshold at two sites in 2009, but did not meet or exceed the lethal threshold at any monitoring location along Whychus Creek in 2010 or in 2011.

The hottest water day in 2011 occurred on August 24th. Temperatures on the hottest water day at all sites monitored in 2011 were lower than or equal to temperatures recorded on the hottest water days from 2007-2010, with the exception of site WC 000.25, where the 2011 temperature exceeded the 2010 temperature by 0.1˚C (Figure 5). The average longitudinal rate of change in 2011 was 0.07˚C per mile, down from the 2007 average rate of 0.1˚C per mile but higher than the 2008, 2009 and 2010 average rates of change (0.03˚C, 0.02˚C, and 0.05˚C per mile, respectively) (Figure 6). The relatively higher average longitudinal rates of change for 2010 and 2011 result from lower temperatures at the upstream-most site, concurrent with later snowmelt runoff and higher flows in these years, 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 generally consistent with previous years' results. 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 (data from WC 024.25 are not available for 2009). High rates of warming also continue to occur at the downstream monitoring stations for both Camp Polk and Rimrock Ranch, channelized sites characterized by high solar insolation that have been prioritized for stream channel restoration projects. As in previous years, the rate of temperature change in Alder Springs reaches was substantially lower than the average rate of change, reflecting the cooling effect of springs complex flows, but less dramatically so than in 2009 and 2010.

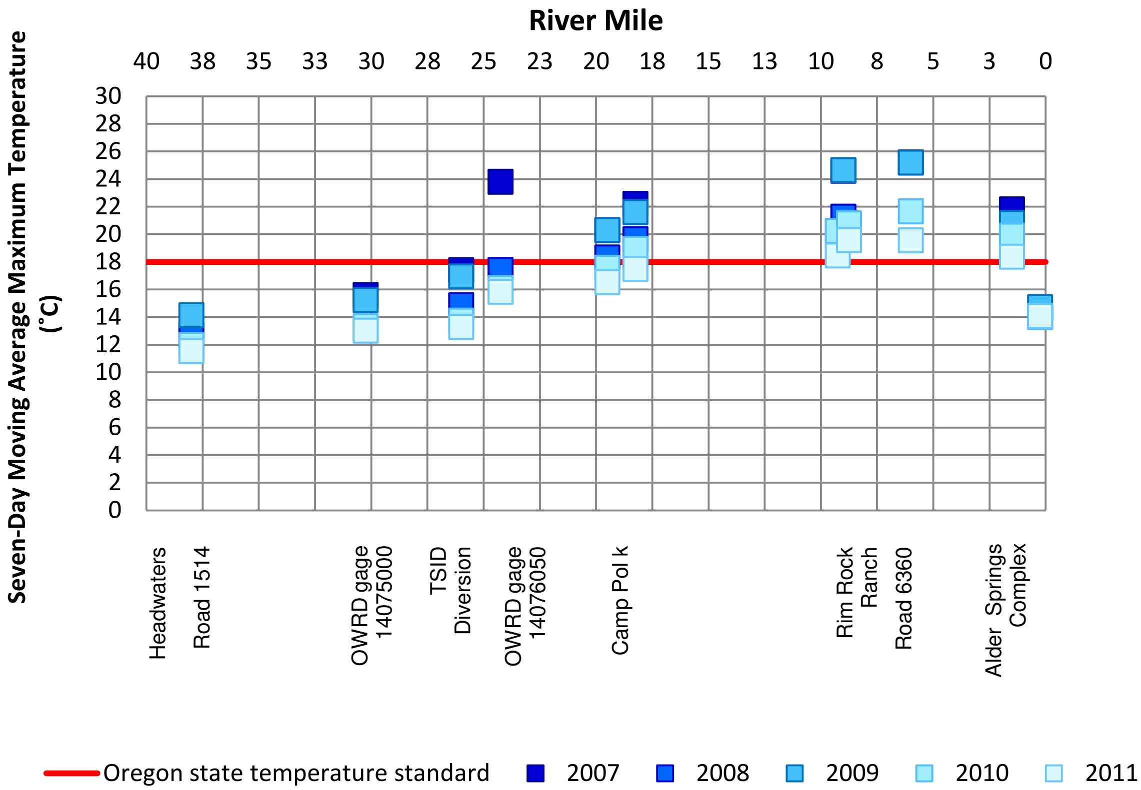


Figure 5.
The longitudinal temperature profile of Whychus Creek on the hottest water days from 2007 through 2011 shows warming from the headwaters to Alder Springs, where springs complex flows cool stream temperatures.

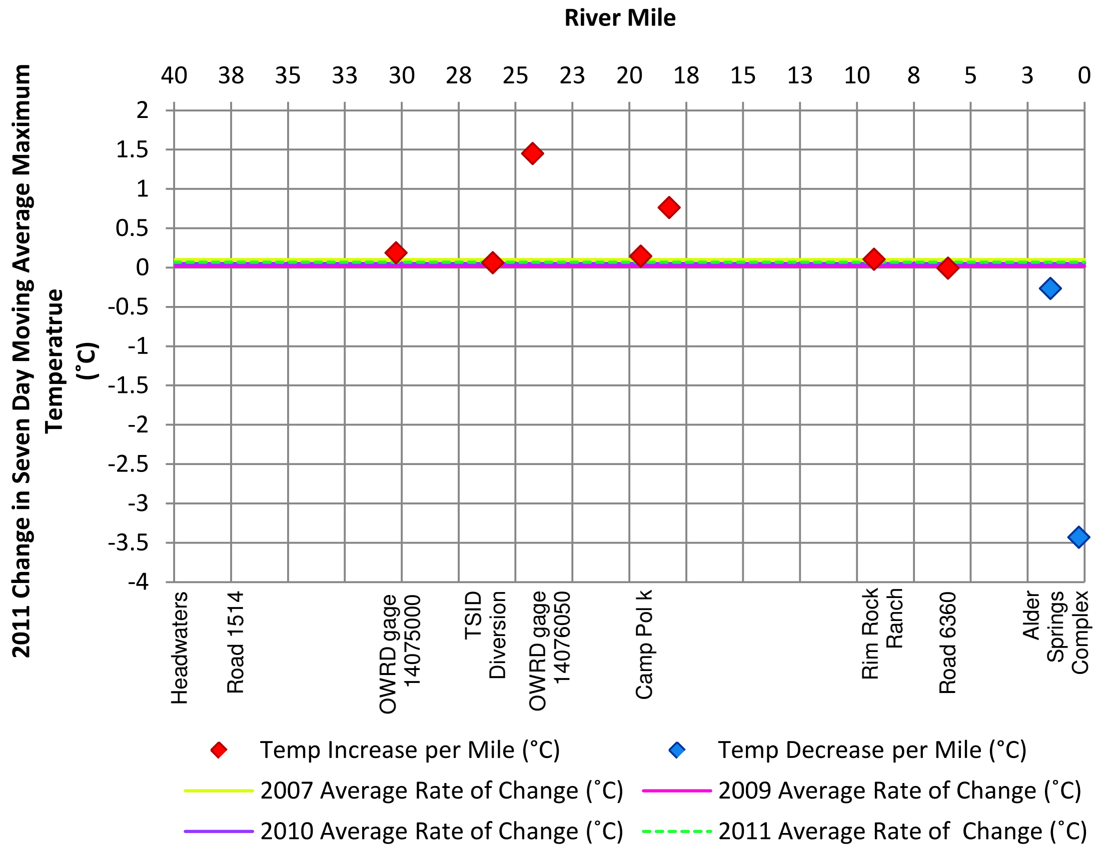


Figure 6.
Longitudinal average rate of temperature change in Whychus Creek in 2011. Higher than average longitudinal changes in temperature identify reaches in which the rate of warming increased and indicate prioritization of these areas for research and restoration planning. Lower than average longitudinal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

Streamflow restoration effectiveness

The 2010-2011 BACI differences of mean July daily median temperatures in restoration reaches 1 and 2 were significantly less than the BACI difference for the reference reach (Table 2, Figure 7), resuming the cooling trend observed in these reaches relative to the reference reach from 2002 to 2006 and from 2002 to 2010. The 2010-2011 BACI difference for restoration reach 3 was significantly greater than the difference for the reference reach, indicating a warming effect in this reach, also in keeping with the 2002-2006 and 2002-2010 trends reported in previous years.

As expected, the BACI difference from 2010 to 2011 for restoration reach 4 tracked the difference observed for restoration reach 2 and was significantly less than the BACI difference for the reference reach, substantiating a cooling response in this reach relative to the reference reach from 2010 to 2011. The mean BACI difference for reach 4 was slightly greater than the difference for reach 2 and therefore slightly warmer than reach 2 relative to the reference reach, possibly due to solar insolation along the 4.5 miles of additional length in reach 4 over reach 2. The difference for restoration reach 5 was substantially greater than the difference for restoration reach 3, isolating the more extreme warming effect of higher, relatively warmer flows from streamflow restoration on the 1.25 miles characterized by cold Alder Springs flows than along the 30 miles included in reach 3.

Table 2. Mean BACI differences and standard deviations for the reference reach and five restoration reaches, and t- and p-values for t-tests comparing the mean for each restoration reach to the mean for the reference reach. Restoration reach means less than the reference reach mean indicate a cooling trend from 2010 to 2011; values that are greater than the reference reach mean indicate a warming trend between data years.

2010-2011			
	mean and sd	t-value	p-value
Reference reach (WC 038.00 - WC 030.25)	**-0.1 ± 0.5**		
Restoration reach 1 (WC 030.25 - WC 024.25)	-0.9 ± 0.7	-5.55	0.00
Restoration reach 2 (WC 030.25 - WC 006.00)	-2.3 ± 1.1	-12.95	0.00
Restoration reach 3 (WC 030.25 - WC 000.25)	0.9 ± 0.8	5.52	0.00
Restoration reach 4 (WC 030.25 - WC 001.50)	-1.8±1.15	-9.3406	0.00
Restoration reach 5 (WC 001.50 - WC 000.25)	2.7±1.13	10.7719	0.00

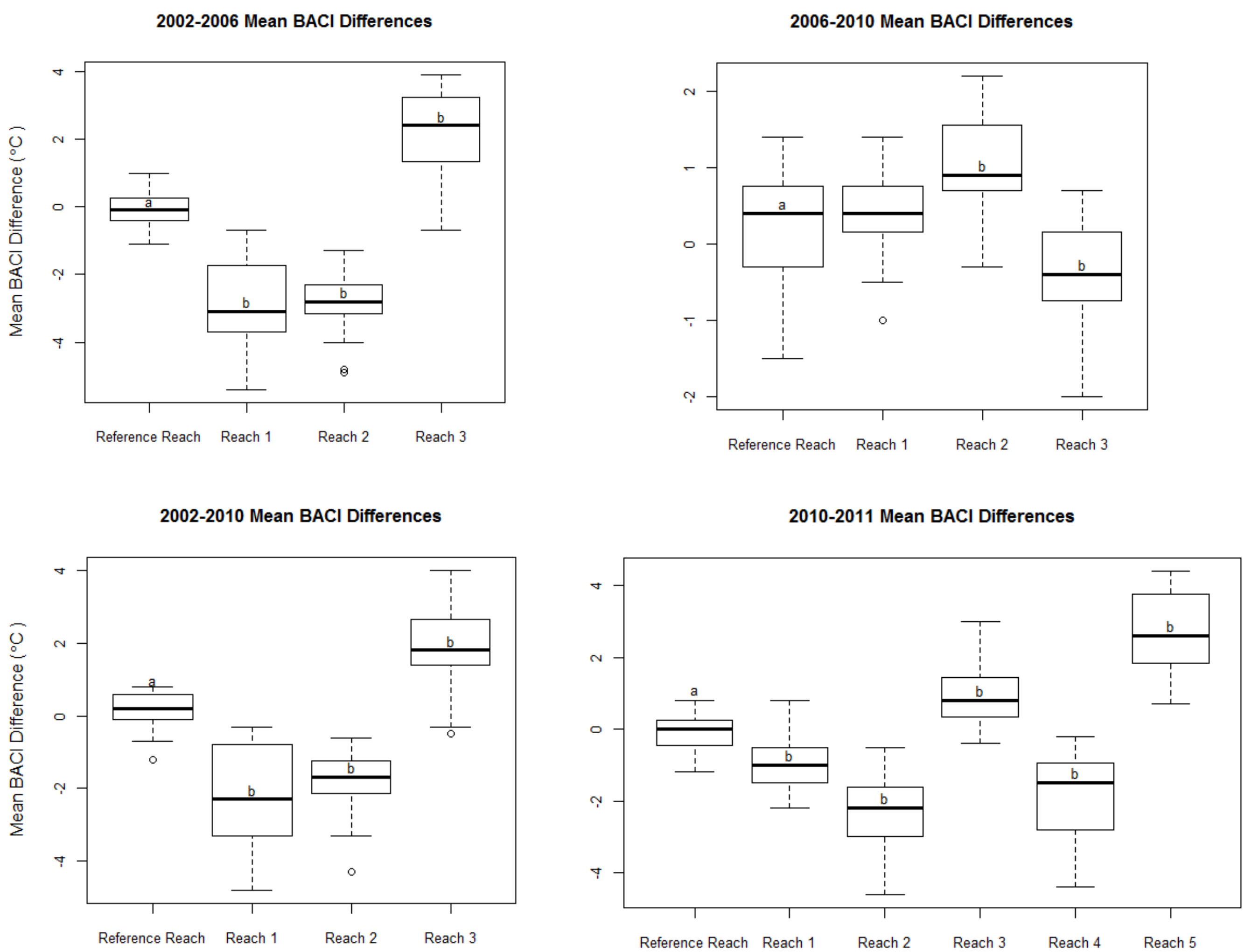


Figure 7.
Mean BACI differences from 2002-2006, 2002-2010, and 2010 to 2011 were significantly less ($p<0.05$) in restoration reaches 1 and 2, and significantly greater in restoration reach 3, than in the reference reach. The mean BACI difference for 2010-2011 reach 4, which duplicates the effects of streamflow in reach 2, was also significantly less than that for the reference reach. The mean difference for 2010-2011 reach 5, which isolates the effect of restored streamflow in the vicinity of Alder Springs, was significantly greater than in the reference reach. From 2006-2010, when actual flows decreased between years, temperature trends reversed, with a significantly greater mean difference in restoration reach 2, indicating warming, and a significantly lower mean difference in restoration reach 3, indicating cooling.

Target streamflow

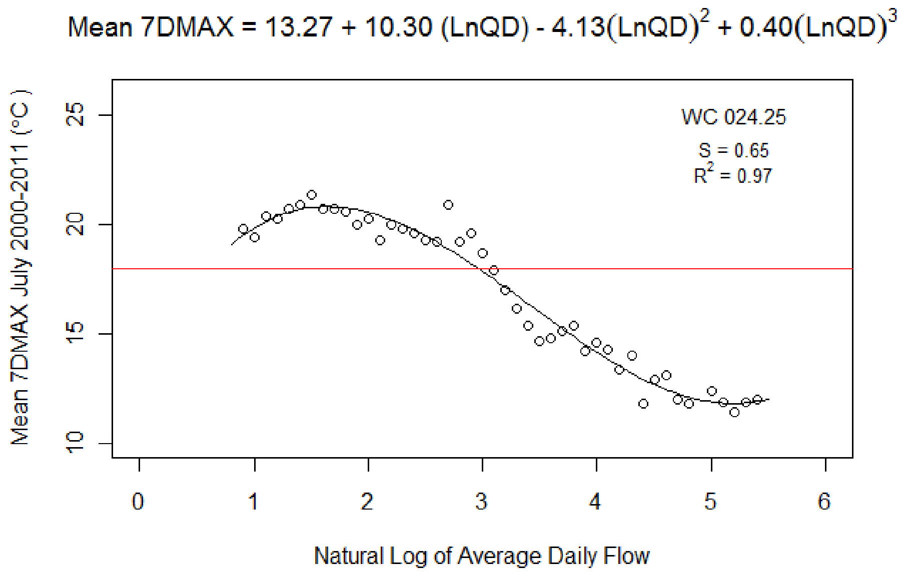
A cubic (third-order polynomial) regression model produced the best fit to 2000-2011 temperature-flow data for both WC 024.25 and WC 006.00 sites (Table 3, Figure 8). Temperatures calculated from the WC 024.25 regression model suggest that 19 cfs (2.9 LnQD) was the minimum streamflow that produced a mean 7DMAX temperature at or below 18°C (± 1.7°C) given temperatures observed from July 2000-2011 at this location. The existing 33 cfs restoration target resulted in a mean 7DMAX temperature of 15.9°C ± 1.7°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 2000-2011 estimate 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 2000-2011 model estimates temperatures meeting the state standard at slightly lower flows than the 21 cfs predicted from earlier (2000-2008; 2000-2010) temperature-flow relationships.

The regression model for temperature-flow relationships at Road 6360 (WC 006.00) derived from 2000-2011 temperature and flow data predicts 66 cfs (4.2 LnQD) to be the minimum streamflow that will achieve a mean 7DMAX temperature of 18.0 ± 2.0°C. Under this model the target streamflow of 33 cfs below Indian Ford Creek is projected to produce a mean 7DMAX temperature of 20.7°C ± 2.0°C, above the 18°C state standard but still below the 24°C lethal temperature threshold. This result is supported by Heat Source model predictions estimating a 7DMAX temperature of 18.5°C ± 1°C at 62 cfs; the 2000-2011 regression model predicts 18.3°C ± 2.0°C at 62 cfs. The 2000-2011 model estimates temperatures meeting the state standard at notably lower flows than the 78 cfs predicted by the 2000-2008 temperature-flow relationship, and is consistent with the 2000-2010 model prediction of 18.0 ± 2.1°C at 66 cfs.

Table 3. A cubic regression provided the best fit to the 2000-2011 temperature-flow data, with the lowest S and highest R^2 values, for both sites. Temperatures calculated using the corresponding regression equation are expected to be the most accurate of the three regression models.

Regression	Equation	df	S	R^2
	WC 24.25 (n=45)			
Linear	24.44-2.44(LnQD)	43	1.121	0.8954
Quadratic	$21.85-0.43(LnQD)-0.32(LnQD)^2$	42	1.006	0.9177
Cubic	$\mathbf{13.27+10.30(LnQD)-4.13(LnQD)^2+0.4(LnQD)^3}$	**41**	**0.6509**	**0.9664**
	WC 06.00 (n=42)			
Linear	28.88-2.67(LnQD)	40	1.589	0.8256
Quadratic	$22.19+2.7(LnQD)-0.89(LnQD)^2$	39	0.9158	0.9435
Cubic	$\mathbf{16.46+10.04(LnQD)-3.57(LnQD)^2+0.29(LnQD)^3}$	**38**	**0.7823**	**0.9598**

a



b

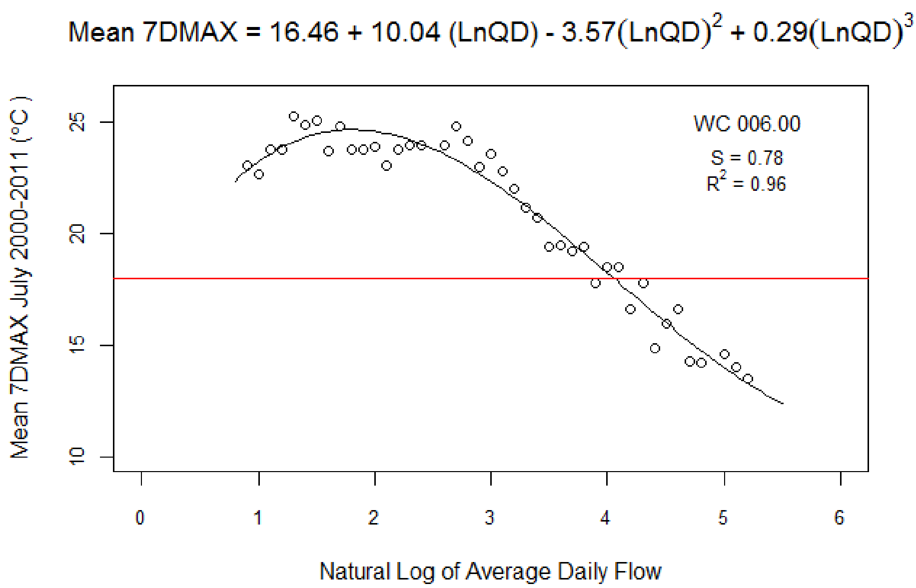


Figure 8.
Regression models fitted to temperature-flow data from July 2000-2011 describe the relationship between temperature and flow observed at **a)** Sister's City Park (WC 024.25) and **b)** Road 6360 (WC 006.00). Corresponding regression equations are used to calculate temperature at a given flow.

Discussion

Temperature status

Temperatures in Whychus Creek in 2011 maintained the cooling trend observed over the last decade while continuing to exceed the state standard at some sites. Although temperatures at Sisters City Park haven't exceeded the 18˚C state standard since 2008, temperatures at Road 6360 (WC 006.00) and at two additional sites (WC 008.75 and WC 001.50) continued to exceeded this criterion in 2011, consistent with the ODEQ 2010 303(d) Category 5 listing of Whychus Creek as water quality limited (ODEQ 2011). While temperatures at Road 6360 exceeded 18˚C during the month of August only, over half of August days at this site were above 18˚C. Notably, site WC 008.75, at the lower end of Rimrock Ranch, surpassed Road 6360 as the site with the most days in excess of the 18˚C standard in 2011.

Because the state standard is biologically based, we can infer that temperature conditions at both WC 006.00 and WC 008.75 were frequently marginal to support salmon and trout rearing and migration. However, despite temperatures consistently exceeding optimal conditions at these sites, temperatures remained below the 24˚C lethal threshold for the second consecutive year. In addition, the highest 7DMAX temperature at these or any site in 2011 was 19.6˚C, barely above the 19.5˚C threshold at which growth rates have been shown to slow in chinook salmon (Brett et al. 1982).

Although temperature conditions for salmon and trout spawning never exceeded the 13˚C biological requirement at WC 024.25 during the January 1 – May 15 spawning season in 2011, 7DMAX temperatures exceeded 13˚C for 29% to 93% of spawning season data days (May 2 to May 15) at five sites between RM 18.25 and RM 1.5. Because data are not available for the spawning season prior to May 2, the proportion of days in excess of the 13˚C biological requirement may be higher than if data were available for more, and earlier, dates. Nonetheless, 7DMAX temperatures for the dates for which data are available indicate suboptimal conditions for salmonid spawning.

While many temperatures exceeding the state standard at Road 6360 (WC 006.00), Rimrock Ranch (WC 008.75), and other sites were recorded at flows lower than the 33 cfs streamflow restoration target, some temperatures in excess of the 18˚C state standard and 13˚C spawning requirement were observed when flows at Sister's City Park were higher than the 33 cfs target downstream of Indian Ford Creek.

Streamflow restoration effectiveness

Changes in temperature in restoration reaches 1, 2, and 4 from 2010 to 2011 substantiate the cooling trend observed from 2002 to 2006 and from 2002 to 2010, and support the hypothesis that increasing streamflow is effective in reducing stream temperature. Temperatures increased in restoration reaches 1 and 2 relative to the reference reach only from 2006 to 2010, corresponding to a decrease in streamflow over the same interval. These results corroborate the relationship between streamflow and temperature described by the temperature-flow regression model.

The temperature response to increased flows in restoration reaches 3 and 5 from 2010 to 2011 was opposite that in the two upstream restoration reaches. These results support the trend observed for all previous intervals analyzed (2002-2006, 2006-2010, and 2002-2010). Restoration reach 3 warmed from 2002 to 2006, 2002 to 2010 and 2010 to 2011 while restoration reaches 1 and 2 cooled, and cooled from 2006 to 2010 while the other restoration reaches warmed. Restoration reach 5, which isolates the counteracting effect of increased warm flows from streamflow restoration on the substantially colder temperatures produced by Alder Springs flows, also warmed significantly from 2010 to 2011. Whereas restoration reaches 1 and 2 receive flows primarily from upstream reaches contingent on annual flow and on the volume of water diverted for irrigation with minimal (5 cfs or less) flow contributions from other sources, reach 3 includes Alder Springs, which contributes an additional 95 cfs of cold water to Whychus flows (UDWC 2008c), and reach 5 encompasses almost exclusively the area where Alder Springs flows enter Whychus. When Whychus flows are low, the coldwater flows of Alder Springs account for a relatively greater proportion of the total streamflow below Alder Springs than when Whychus flows are high and Alder Springs flows contribute a relatively smaller proportion of the total volume below the springs complex.

Although baseline flows from Alder Springs have been estimated, there is no permanent stream gage or other ongoing monitoring to measure changes in the contribution of Alder Springs to Whychus Creek. 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). The substantial influence of Alder Springs flows on Whychus Creek temperature in restoration reaches 3 and 5 suggested by the divergent trends in these reaches for all intervals analyzed 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. 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.

While inclusion of Whychus Creek river miles 0.25 to 1.50 (WC 000.25 – WC 001.50) and therefore Alder Springs flows in restoration reach 3 has provided important preliminary information about the influence of Alder Springs on Whychus Creek temperatures, including this reach also confounds the longitudinal effects of restored streamflow on temperature and compromises the utility of this analysis to evaluate the extent to which streamflow restoration is effective in reducing temperature. The 2010-2011 analysis of river miles 30.25 to 0.25 (WC 030.25 – 000.25) as two separate reaches above and below the coldwater contribution of Alder Springs, in previous years analyzed as a single restoration reach, demonstrates a strong cooling response to increased flows between WC 030.25, immediately upstream of the point of major irrigation diversions, and WC 001.50, immediately upstream of Alder Springs. Analysis of river miles 1.50 to 0.25 as a distinct reach removes the interaction of cold Alder Springs flows and warm restoration flows from the upstream response to streamflow restoration, and illustrates the warming effect of restored flows in that specific location. It is important to note that the warming effect observed in restoration reaches 3 and 5 is a measure of the response to restoration flows in these reaches relative to changes in temperature over the same interval in the reference reach, and does not represent a net warming trend in the Alder Springs area.

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 2000-2011 temperature-flow model predicts temperatures of 15.9°C ±1.7°C at 33 cfs at Sisters City Park, well below the state temperature standard and salmonid rearing and migration requirements, the estimate of 20.7°C ±2°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. The temperature-flow relationship observed from 2000 to 2011 suggests that the 18°C state temperature standard will only be predictably met at Road 6360 at minimum flows of 66 cfs at Sisters City Park.

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 flows supported up to 20 spawning chinook salmon or redds counted in the creek in 1952, and steelhead 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. Given the present flow target of 33 cfs below Indian Ford Creek, we expect 7DMAX temperatures to consistently exceed the 18°C state standard at Road 6360 (WC 006.00) in July, upholding the 303(d) listing of Whychus Creek as temperature-impaired. However, 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 20.7°C 7DMAX predicted at this level 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.

Conclusions

Temperatures observed in Whychus Creek continue to chart a cooling trend as flows increase with streamflow restoration. Whychus temperatures were lower on the hottest water day in 2011 than in 2010 and lower in 2010 than in 2009. The average rate of longitudinal temperature change fell from 2007 to 2009. While the average rate of change increased from 2009 to 2010 and from 2010 to 2011, this increase represents lower hottest water day temperatures at the upstream-most sites relative to downstream sites. As of 2011 7DMAX temperatures continued to exceed state temperature standards for salmon and trout rearing and migration between RM 8.75 and RM 1.5 and for salmon and trout spawning between RM 18.25 and 1.5, but at fewer sites than in 2010, and 7DMAX temperatures did not meet or exceed the 24°C lethal threshold at any site for the second year in a row. For every year included in analysis, when flows have increased between years, temperatures in reaches with restored flows above Alder Springs cooled more than temperatures in the reference reach representing natural flow conditions, in direct response to streamflow restoration.

The temperature-flow relationship described by eleven 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), the hottest location for which temperature data are available, 66 cfs is the minimum flow estimated to meet the 18°C standard. Although 66 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 within Whychus Canyon Preserve (RM 9 - 15), will 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 demonstrate that streamflow restoration has already improved temperature conditions for re-introduced salmon and trout, and expand 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
WC 038.00	Road 1514				TE				TE RE				TE RE		TE	TE	TE RE	TE RE
WC 030.25	OWRD Gage 14075000					TE			TE RE	TE	TE	TE	TE RE	TE		TE	TE RE	TE RE
WC 026.00	Road 4606 Footbridge					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
WC 019.50	d/s Camp Polk bridge				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
WC 009.25	u/s end of Rimrock Ranch															TE	TE	TE
WC 009.00	Rimrock Ranch												TE	TE	TE			
WC 008.75	d/s end of Rimrock Ranch															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
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
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

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	18.6	1.7	**57**	13.8	1.6	**112**	11.9	1.6	**167**	11.4	1.6
3	20.1	1.8	**58**	13.8	1.6	**113**	11.9	1.6	**168**	11.4	1.6
4	20.7	1.8	**59**	13.7	1.6	**114**	11.9	1.6	**169**	11.4	1.6
5	20.8	1.8	**60**	13.7	1.6	**115**	11.9	1.6	**170**	11.4	1.6
6	20.8	1.8	**61**	13.6	1.6	**116**	11.9	1.6	**171**	11.4	1.6
7	20.6	1.8	**62**	13.6	1.6	**117**	11.9	1.6	**172**	11.4	1.6
8	20.4	1.8	**63**	13.5	1.6	**118**	11.8	1.6	**173**	11.4	1.6
9	20.2	1.8	**64**	13.4	1.6	**119**	11.8	1.6	**174**	11.4	1.6
10	20.0	1.8	**65**	13.4	1.6	**120**	11.8	1.6	**175**	11.4	1.6
11	19.7	1.8	**66**	13.3	1.6	**121**	11.8	1.6	**176**	11.4	1.6
12	19.5	1.8	**67**	13.3	1.6	**122**	11.8	1.6	**177**	11.4	1.6
13	19.3	1.8	**68**	13.2	1.6	**123**	11.8	1.6	**178**	11.4	1.6
14	19.0	1.7	**69**	13.2	1.6	**124**	11.8	1.6	**179**	11.4	1.6
15	18.8	1.7	**70**	13.2	1.6	**125**	11.7	1.6	**180**	11.4	1.6
16	18.6	1.7	**71**	13.1	1.6	**126**	11.7	1.6	**181**	11.4	1.6
17	18.4	1.7	**72**	13.1	1.6	**127**	11.7	1.6	**182**	11.4	1.6
18	18.2	1.7	**73**	13.0	1.6	**128**	11.7	1.6	**183**	11.4	1.6
19	18.0	1.7	**74**	13.0	1.6	**129**	11.7	1.6	**184**	11.4	1.6
20	17.8	1.7	**75**	12.9	1.6	**130**	11.7	1.6	**185**	11.4	1.6
21	17.6	1.7	**76**	12.9	1.6	**131**	11.7	1.6	**186**	11.4	1.6
22	17.5	1.7	**77**	12.9	1.6	**132**	11.7	1.6	**187**	11.4	1.6
23	17.3	1.7	**78**	12.8	1.6	**133**	11.7	1.6	**188**	11.4	1.6
24	17.1	1.7	**79**	12.8	1.6	**134**	11.6	1.6	**189**	11.4	1.6
25	17.0	1.7	**80**	12.8	1.6	**135**	11.6	1.6	**190**	11.4	1.6
26	16.8	1.7	**81**	12.7	1.6	**136**	11.6	1.6	**191**	11.4	1.6
27	16.7	1.7	**82**	12.7	1.6	**137**	11.6	1.6	**192**	11.4	1.6
28	16.5	1.7	**83**	12.7	1.6	**138**	11.6	1.6	**193**	11.4	1.6
29	16.4	1.7	**84**	12.6	1.6	**139**	11.6	1.6	**194**	11.4	1.6
30	16.3	1.7	**85**	12.6	1.6	**140**	11.6	1.6	**195**	11.4	1.6
31	16.1	1.7	**86**	12.6	1.6	**141**	11.6	1.6	**196**	11.4	1.6
32	16.0	1.7	**87**	12.5	1.6	**142**	11.6	1.6	**197**	11.4	1.6
33	15.9	1.7	**88**	12.5	1.6	**143**	11.6	1.6	**198**	11.4	1.6
34	15.8	1.7	**89**	12.5	1.6	**144**	11.6	1.6	**199**	11.4	1.6
35	15.7	1.7	**90**	12.4	1.6	**145**	11.5	1.6	**200**	11.4	1.6
36	15.6	1.7	**91**	12.4	1.6	**146**	11.5	1.6	**201**	11.4	1.6
37	15.4	1.7	**92**	12.4	1.6	**147**	11.5	1.6	**202**	11.4	1.6
38	15.3	1.7	**93**	12.4	1.6	**148**	11.5	1.6	**203**	11.4	1.6
39	15.2	1.7	**94**	12.3	1.6	**149**	11.5	1.6	**204**	11.4	1.6
40	15.1	1.7	**95**	12.3	1.6	**150**	11.5	1.6	**205**	11.4	1.6
41	15.0	1.7	**96**	12.3	1.6	**151**	11.5	1.6	**206**	11.4	1.6
42	15.0	1.7	**97**	12.3	1.6	**152**	11.5	1.6	**207**	11.4	1.6
43	14.9	1.7	**98**	12.2	1.6	**153**	11.5	1.6	**208**	11.4	1.6
44	14.8	1.7	**99**	12.2	1.6	**154**	11.5	1.6	**209**	11.4	1.6
45	14.7	1.7	**100**	12.2	1.6	**155**	11.5	1.6	**210**	11.4	1.6
46	14.6	1.7	**101**	12.2	1.6	**156**	11.5	1.6	**211**	11.4	1.6
47	14.5	1.7	**102**	12.1	1.6	**157**	11.5	1.6	**212**	11.4	1.6
48	14.5	1.7	**103**	12.1	1.6	**158**	11.5	1.6	**213**	11.4	1.6
49	14.4	1.7	**104**	12.1	1.6	**159**	11.5	1.6	**214**	11.4	1.6
50	14.3	1.7	**105**	12.1	1.6	**160**	11.5	1.6	**215**	11.4	1.6
51	14.2	1.7	**106**	12.1	1.6	**161**	11.5	1.6	**216**	11.4	1.6
52	14.2	1.7	**107**	12.0	1.6	**162**	11.4	1.6	**217**	11.4	1.6
53	14.1	1.7	**108**	12.0	1.6	**163**	11.4	1.6	**218**	11.4	1.6
54	14.0	1.7	**109**	12.0	1.6	**164**	11.4	1.6	**219**	11.4	1.6
55	14.0	1.6	**110**	12.0	1.6	**165**	11.4	1.6	**220**	11.4	1.6
56	13.9	1.6	**111**	12.0	1.6	**166**	11.4	1.6	**221**	11.4	1.6

Whychus Creek at Road 6360 (WC 006.00) at flows from 2.4 cfs to 190 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	21.8	2.1	**57**	18.6	2.0	**112**	16.2	1.9	**167**	15.0	1.9
3	23.6	2.1	**58**	18.5	2.0	**113**	16.1	1.9	**168**	15.0	1.9
4	24.3	2.1	**59**	18.5	2.0	**114**	16.1	1.9	**169**	15.0	1.9
5	24.6	2.1	**60**	18.4	2.0	**115**	16.1	1.9	**170**	15.0	1.9
6	24.6	2.1	**61**	18.3	2.0	**116**	16.1	1.9	**171**	15.0	1.9
7	24.6	2.1	**62**	18.3	2.0	**117**	16.0	1.9	**172**	15.0	1.9
8	24.5	2.1	**63**	18.2	2.0	**118**	16.0	1.9	**173**	14.9	1.9
9	24.3	2.1	**64**	18.2	2.0	**119**	16.0	1.9	**174**	14.9	1.9
10	24.2	2.1	**65**	18.1	2.0	**120**	16.0	1.9	**175**	14.9	1.9
11	24.0	2.1	**66**	18.0	2.0	**121**	15.9	1.9	**176**	14.9	1.9
12	23.8	2.1	**67**	18.0	2.0	**122**	15.9	1.9	**177**	14.9	1.9
13	23.7	2.1	**68**	17.9	2.0	**123**	15.9	1.9	**178**	14.9	1.9
14	23.5	2.1	**69**	17.9	2.0	**124**	15.9	1.9	**179**	14.9	1.9
15	23.3	2.1	**70**	17.8	2.0	**125**	15.8	1.9	**180**	14.9	1.9
16	23.1	2.1	**71**	17.8	2.0	**126**	15.8	1.9	**181**	14.8	1.9
17	22.9	2.1	**72**	17.7	2.0	**127**	15.8	1.9	**182**	14.8	1.9
18	22.8	2.1	**73**	17.7	2.0	**128**	15.8	1.9	**183**	14.8	1.9
19	22.6	2.1	**74**	17.6	2.0	**129**	15.7	1.9	**184**	14.8	1.9
20	22.5	2.1	**75**	17.6	2.0	**130**	15.7	1.9	**185**	14.8	1.9
21	22.3	2.1	**76**	17.5	2.0	**131**	15.7	1.9	**186**	14.8	1.9
22	22.1	2.1	**77**	17.5	2.0	**132**	15.7	1.9	**187**	14.8	1.9
23	22.0	2.1	**78**	17.4	2.0	**133**	15.6	1.9	**188**	14.8	1.9
24	21.8	2.1	**79**	17.4	2.0	**134**	15.6	1.9	**189**	14.7	1.9
25	21.7	2.1	**80**	17.3	2.0	**135**	15.6	1.9	**190**	14.7	1.9
26	21.6	2.1	**81**	17.3	2.0	**136**	15.6	1.9			
27	21.4	2.1	**82**	17.2	2.0	**137**	15.6	1.9			
28	21.3	2.1	**83**	17.2	2.0	**138**	15.5	1.9			
29	21.2	2.1	**84**	17.2	2.0	**139**	15.5	1.9			
30	21.1	2.1	**85**	17.1	2.0	**140**	15.5	1.9			
31	20.9	2.0	**86**	17.1	2.0	**141**	15.5	1.9			
32	20.8	2.0	**87**	17.0	2.0	**142**	15.5	1.9			
33	20.7	2.0	**88**	17.0	2.0	**143**	15.4	1.9			
34	20.6	2.0	**89**	17.0	1.9	**144**	15.4	1.9			
35	20.5	2.0	**90**	16.9	1.9	**145**	15.4	1.9			
36	20.4	2.0	**91**	16.9	1.9	**146**	15.4	1.9			
37	20.3	2.0	**92**	16.8	1.9	**147**	15.4	1.9			
38	20.2	2.0	**93**	16.8	1.9	**148**	15.3	1.9			
39	20.1	2.0	**94**	16.8	1.9	**149**	15.3	1.9			
40	20.0	2.0	**95**	16.7	1.9	**150**	15.3	1.9			
41	19.9	2.0	**96**	16.7	1.9	**151**	15.3	1.9			
42	19.8	2.0	**97**	16.7	1.9	**152**	15.3	1.9			
43	19.7	2.0	**98**	16.6	1.9	**153**	15.3	1.9			
44	19.6	2.0	**99**	16.6	1.9	**154**	15.2	1.9			
45	19.5	2.0	**100**	16.6	1.9	**155**	15.2	1.9			
46	19.4	2.0	**101**	16.5	1.9	**156**	15.2	1.9			
47	19.3	2.0	**102**	16.5	1.9	**157**	15.2	1.9			
48	19.3	2.0	**103**	16.5	1.9	**158**	15.2	1.9			
49	19.2	2.0	**104**	16.4	1.9	**159**	15.2	1.9			
50	19.1	2.0	**105**	16.4	1.9	**160**	15.1	1.9			
51	19.0	2.0	**106**	16.4	1.9	**161**	15.1	1.9			
52	19.0	2.0	**107**	16.3	1.9	**162**	15.1	1.9			
53	18.9	2.0	**108**	16.3	1.9	**163**	15.1	1.9			
54	18.8	2.0	**109**	16.3	1.9	**164**	15.1	1.9			
55	18.7	2.0	**110**	16.2	1.9	**165**	15.1	1.9			
56	18.7	2.0	**111**	16.2	1.9	**166**	15.0	1.9			

Habitat Quality in Whychus Creek

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Abstract

Human actions have altered stream habitat in Whychus Creek, a tributary to Oregon's Deschutes River, for over 100 years. The Oregon Department of Fish and Wildlife surveyed physical habitat in the creek in 1997 and in 2008/2009/2011 in support of an anadromous fish reintroduction effort. Survey data were used in the HabRate model to develop reach-scale habitat ratings for the creek by anadromous species and life stage. Reaches assigned during 2008-2009 sampling did not correspond spatially to those assigned and sampled in 1997, resulting in reach-level ratings that described different spatial extents and limited comparison across sampling periods. UDWC partnered with ODFW to re-assign 1997 and 2008-2009 survey data to new reaches encompassing the same geographic extents and generate ratings for the new reaches using an updated version of the HabRate model. The resulting ratings varied by species and by life stage. Ratings indicated that habitat in 1997 was best for age 0+ overwintering steelhead, and suitable for all rearing life stages of steelhead and chinook. Ratings indicated habitat was unsuitable for steelhead and chinook spawning in all but 2.7 miles of Whychus Creek. Ratings for 2008/2009/2011 indicated habitat was best for age 1+ summer rearing steelhead and for rearing chinook, and was suitable for all rearing life stages for both species. Habitat was suitable for steelhead and chinook spawning in 2008/2009/2011 in 22.7 miles but remained unsuitable in 10.5 miles. From 1997 to 2008/2009, habitat quality ratings declined for summer rearing and overwintering stages of steelhead and chinook in 0.54 to 1.57 miles and improved for age 0+ winter chinook in 11.4 miles. Habitat quality ratings declined for steelhead and chinook spawning in 1.57 miles, but improved in 11.4 miles. Improvements in habitat quality ratings resulted from recorded decreases in fine sediment; declines in ratings were a result of increases in fine sediments and decreases in boulders and undercut banks. Given that no channel habitat restoration occurred in reaches where habitat attributes and consequently ratings changed between years, no clear explanation exists for the changes observed. Future analyses comparing habitat quality in restored reaches to 2008/2009/2011 conditions will help restoration partners understand how habitat conditions in Whychus Creek change following restoration activities.

Introduction

Human actions over the last century, including irrigation diversions, channel straightening, riparian grazing and other actions, have altered the physical structure of Whychus Creek, limiting stream functions and affecting riparian and aquatic habitat quality. According to NRC (2002), habitat alteration occurs through human disturbances or the prevention of natural disturbances. Both of these types of events have occurred in Whychus Creek.

Habitat alteration may improve or degrade habitat quality (NRC 2002). The specific outcomes of habitat alteration depend on the type of alteration and the stream system altered. Decreased riparian cover may lead to warmer stream temperatures (Poole and Berman 2000) and warmer stream temperatures may affect salmonid spawning timing and egg mortality (Richter and Kolmes 2005). In contrast, other alterations may lead to increased production for some species (NRC 2002). Variation in salmonid life history strategies makes it difficult to generalize about the effects of habitat alteration on specific populations.

Local restoration partners have explicitly identified the goal of providing the habitat necessary to support naturally reproducing resident and anadromous fish populations in Whychus Creek. They have recognized that, although they cannot manage fish populations or conditions outside of the creek, they can improve opportunities for survival and growth within the creek. Given this goal, habitat quality stands out as an indicator of restoration effectiveness in Whychus Creek.

Restoration partners expect that their actions, ranging from channel reconstruction to stream flow restoration, will improve conditions for anadromous fish. They selected the Oregon Department of Fish and Wildlife's HabRate model (Burke *et al* 2010) as one tool to document changes in habitat conditions that occur in Whychus Creek. HabRate assigns literature-based ratings to stream survey data to develop an aggregate reach-level rating for each life stage of each species of interest. It provides the basis for the habitat ratings appearing in this study.

In 1997 Oregon Department of Fish and Wildlife surveyed physical habitat quality along approximately 27 miles of Whychus Creek to establish baseline habitat quality data to inform the planned reintroduction of anadromous steelhead and salmon. From 2008 to 2009 ODFW re-surveyed the original 27 miles as well as five miles that hadn't been surveyed in 1997. Surveys quantified numerous parameters identified in peer-reviewed literature as important characteristics determining habitat suitability for spawning and emergence and rearing and migration salmonid life stages.

In 2009 ODFW used the HabRate 2.0 model (Burke *et al* 2003) to assign ratings to each parameter and aggregate ratings to the unit and reach level to produce a reach-level rating of habitat suitability for each reach for each sampling period (1997 and 2008/2009) based on parameter values for the five freshwater life stages of steelhead trout and the three freshwater life stages of spring Chinook salmon. UDWC used ODFW ratings to conduct a baseline analysis of habitat quality in 1997, prior to initial restoration efforts, and in 2008/2009, following extensive streamflow restoration, to quantify changes in habitat quality over the 11- to 12- year interval between sampling periods (Golden 2010). Because sampling reaches and units were assigned in the field on the basis of numerous factors including land ownership, topography, and channel morphology, and stream access to conduct surveys varied between sampling periods, 2008/2009 sampling reaches did not spatially correspond to 1997 sampling reaches. Consequently, a given length of creek and the parameter values describing it may have been assigned to one reach in 1997 and an adjacent reach during 2008/2009 sampling. Changes in habitat quality from 1997 to 2008/2009 presented by Golden (2010) were therefore based on a comparison of habitat ratings generated from habitat survey data for different spatial and geographic extents and are inherently less precise and less accurate than a direct comparison of ratings based on habitat survey data from the exact same section of stream in 1997 and in 2008/2009.

In 2011 ODFW was granted access to survey an additional 1.35 miles that had not been surveyed in either 1997 or from 2008 to 2009. UDWC partnered with ODFW to create new, spatially aligned 1997 and 2008/2009/2011 reaches from 1997 and 2008/2009/2011 ODFW and USFS habitat survey data spanning 23.6 miles of Whychus Creek. ODFW used the updated HabRate 3.1 model (Burke *et al* 2010) to generate new habitat quality ratings for the resulting reaches. UDWC repeated the 2010 analysis

(Golden 2010) of habitat suitability in Whychus Creek for each freshwater anadromous life stage and change in habitat suitability for each life stage from 1997 to 2011.

Methods

Data Collection

This analysis includes habitat survey data collected from Whychus Creek in 1997 and in 2008/2009. It does not include Indian Ford Creek or Pole Creek, both tributaries to Whychus Creek. These creeks historically connected to Whychus Creek. Indian Ford Creek typically dries up before reaching Whychus Creek under current conditions and Pole Creek had been diverted away from its mouth and into a new channel through 2011.

<u>1997 Data</u>

Oregon Department of Fish and Wildlife (ODFW) and the United States Forest Service (USFS) surveyed Whychus Creek in 1997. ODFW surveyed from the mouth of the creek upstream to the Three Sisters Irrigation District Diversion. They surveyed eight reaches covering 15.2 miles. Landowners did not grant access to an additional four reaches covering 7.2 miles within this section of creek. USFS surveyed the creek upstream from the Three Sisters Irrigation District Diversion (Spateholts 2009a). They surveyed an additional five reaches covering 12.8 miles.

According to Burke *et al* (2003), surveyors based their survey methodology on Hankin (1984) and Hankin and Reeves (1988). Both the Forest Service and ODFW used modified versions of these survey methodologies. Their methodology varied slightly between years and agencies. Their qualitative and quantitative data remained consistent even with these variations (Burke *et al* 2003). Burke *et al* (2003) gathered available 1997 ODFW and USFS survey data and compiled it into reach level data for use in the HabRate model.

<u>2008/2009/2011 Data</u>

Portland General Electric and the Confederated Tribes of the Warm Springs Reservation (Licensees) contracted with ODFW's Aquatic Inventory Project to survey Whychus Creek in 2008. ODFW surveyed the creek from July through September of 2008. They surveyed a total of 22.6 miles in sixteen reaches from the mouth of the creek upstream to the Plainview Ditch (ODFW 2008). Landowners did not grant access to four reaches covering 2.6 miles. In 2009, licensees contracted with ODFW to survey the remaining reaches of Whychus Creek between the Plainview Ditch and Whychus Creek Falls. In 2011 ODFW was granted permission to conduct surveys on an additional 1.7 miles of the 2.6 miles not surveyed in 2008.

ODFW followed standard survey methods in Whychus Creek (ODFW 2006). Surveyors identified the channel form, valley form, streamside vegetation characteristics, water temperature, stream flow, land use, and location for each reach. They further divided each reach into channel habitat units based on bedform, gradient, and substrate (ODFW 2006). Within each habitat unit, survey crews identified the channel form, channel characteristics, wood presence, and riparian conditions (ODFW 2006). ODFW compiled 2008/2009/2011 survey data into reach-level data for use in the HabRate model.

Data Analysis

Model selection

UDWC's baseline analysis (Golden 2010) referenced habitat quality ratings generated from ODFW Aquatic Inventories Project habitat survey data using the HabRate habitat quality model (Burke *et al* 2003, Burke *et al* 2010). Prior to conducting the new analysis of habitat quality in Whychus Creek UDWC evaluated the Unit Characteristic Method as a potential alternative model to HabRate. Whereas HabRate rates habitat quality for a given life stage and species on the basis of observed habitat attribute values according to fish habitat requirements identified in the scientific literature, the Unit Characteristic Method (UCM) predicts the O. mykiss capacity of a stream as a function of the degree to which habitat attributes in a given habitat unit type vary from expected values, given a standard fish density assigned for those expected values. The UCM thus incorporates survey data to predict fish density but does not assign a value or rating to habitat quality separately from the fish density estimate. Given these differences in model design, output, and intended use, UDWC selected HabRate as the more appropriate model for the specific application of quantifying and evaluating habitat quality and changes in habitat quality over time.

Under the 2005 FERC operating license (FERC 2005) for the Pelton-Round Butte Hydroelectric Project (PRB), PGE and the Confederated Tribes of Warm Springs survey approximately 20 miles of stream upstream of PRB every year and use survey data in both HabRate and the UCM to monitor habitat effectiveness and riparian conditions, and to estimate production capacity (FERC 2005). UCM production capacity estimates for Whychus Creek generated from the same ODFW 2008/2009 data we analyze here are available in PGE's 2008 and 2009 Native Fish Monitoring Habitat Component reports (Spateholts 2009b, Spateholts 2010).

ODFW developed the HabRate model to evaluate stream habitat suitability for chinook and sockeye salmon and steelhead trout in the Deschutes Basin. HabRate developers used a combination of literature reviews and professional judgment to identify the habitat attributes and values necessary to support each life history stage of each species (Burke *et al* 2010). The model applies these criteria to assign ratings of poor, fair or good to observed habitat attribute values for each life stage of each species. Level 1 habitat attribute ratings for each life stage of each species (e.g. percent fines, pool area) are weighted and summarized into Level 2 reach attributes (e.g. substrate, morphology) to represent a collective condition for Level 2 attributes. Level 2 attribute ratings are aggregated to produce a reach-level rating of poor, fair, or good for each species and life stage. Ratings of fair and good indicate adequate and optimal habitat conditions, respectively, representing suitable habitat that will support fish survival; a poor rating flags unsuitable habitat conditions that will not support fish survival.

In January 2011 ODFW released HabRate version 3.1 (Burke et al. 2010), which updated the earlier HabRate 2.0 model to reflect the most current understanding of fish habitat needs in the Deschutes Basin. Whereas HabRate 2.0 included measures related to streamflow and temperature in calculations of reach-level ratings, the 3.1 model does not include flow or temperature metrics in calculating reach-level ratings. Habitat quality ratings generated by HabRate 3.1 and presented here thus incorporate only physical habitat measures and remove the influence of flow, which is subject to frequent and rapid fluctuations and changes independently of physical channel structure, and of temperature, which is strongly correlated with flow.

Reach Alignment

2008-2009 habitat surveys assigned habitat units and reaches that differed in spatial extent and location from those assigned during 1997 surveys. The resulting reaches and HabRate reach-level habitat quality ratings for 1997 and for 2008/2009 data were thus not directly comparable. In 2009 UDWC performed a preliminary analysis with existing reaches and available HabRate habitat quality ratings (Golden, 2010). UDWC merged the 1997 and 2008/2009 HabRate data in a new GIS dataset with additional reach divisions that aligned 1997 and 2008/2009 reaches that were surveyed in both years. Each new reach contained 1997 and 2008/2009 habitat ratings. Reaches that were not surveyed in one or both years were reported as unsurveyed. UDWC used this new data set to determine the extent of habitat changes between 1997 and 2008/2009. Changes in reach conditions were classified as improved, unchanged, declined, or unsurveyed based on changes in habitat ratings between 1997 and 2008/2009. UDWC quantified the changes in habitat ratings for each life history stage of each species based on GIS determined reach lengths.

Beginning in 2010 UDWC partnered with ODFW to re-assign 1997 and 2008/2009 habitat survey data to new 1997 and 2008/2009 reaches that as closely as possible spanned the same spatial extents. UDWC obtained the 1997, 2008/2009, and 2011 HabRate datasets from ODFW. The new 2011 data were included with the 2008/2009 dataset for reporting 2008/2009/2011 habitat quality ratings. But, because the reach surveyed in 2011 had not been surveyed in 1997, the 2011 reach was not included in the comparison of 1997 and 2008/2009 data. UDWC used GIS to compare the geographic location of 1997 and 2008/2009 channel habitat units, the smallest unit at which data were recorded, and established new reach boundaries to group units where a) data existed for both 1997 and 2008/2009, b) data existed for only 1997 or 2008/2009, or c) to isolate reaches that were surveyed in neither 1997 nor in 2008/2009.

ODFW used the new reaches in HabRate 3.1 model runs to generate new habitat quality ratings for 1997 and 2008/2009/2011 data. UDWC tallied reach lengths measured in GIS to quantify miles of habitat rated as good, fair, poor, or unsurveyed for each life history stage of each species for all surveyed reaches. We were unable to obtain the original 1997 habitat survey data for Whychus Creek above the Three Sisters Irrigation Diversion, and were therefore unable to generate habitat quality ratings using the revised HabRate model or to align 1997 and 2008/2009 reaches above TSID. We compared new 1997 and 2008/2009 ratings to refine our analysis of changes in habitat quality from 1997-2009 and categorized changes in reach conditions as improved, unchanged, declined, or unsurveyed. We referenced the HabRate model to identify the changes in habitat attributes driving differences in habitat quality ratings between years.

Results

Re-alignment of 1997 and 2008/2009/2011 channel habitat unit data resulted in 21 reaches totaling 23.6 miles from the mouth of Whychus Creek to TSID (Table 1). Of the 23.6 miles, 16.1 were surveyed in both 1997 and from 2008 to 2011. Habitat data were available for 16.3 miles surveyed by ODFW in 1997, with 7.3 miles between the mouth of Whychus Creek and TSID unsurveyed. Data were available for a total of 32.3 miles surveyed between 2008 and 2011, with 0.9 miles remaining unsurveyed between the mouth of Whychus and Whychus Falls. Because the GIS data associated with the HabRate ratings was created from a different source than the GIS data used to measure Whychus Creek river miles, lengths reported for habitat surveys and ratings are different than river miles.

Table 1. Reaches and lengths surveyed in 1997 and in 2008/2009/2011, and re-aligned reaches and lengths representing habitat survey data from the same geographic extent in 1997 and 2008/2009.

	1997	2008/2009/2011	1997 and 2008/2009
Number of Reaches	21	25	21
Extent of surveys (Total length, mi)	23.6	33.2	23.6
Length surveyed (mi)	16.3	32.3	16.1
Length unsurveyed (mi)	7.3	0.9	7.5

Revised calculation of 2008/2009/2011 reach-level habitat quality ratings for the new reaches using the HabRate 3.1 model, presented in this report, resulted in overall higher ratings for steelhead and chinook than ratings generated under the HabRate 2.0 model and presented in the 2009 Habitat Quality in Whychus Creek report (Golden 2010). Although differences in ratings produced by the 2.0 and 3.1 HabRate models were influenced by numerous changes in the attributes included and the relative importance of those attributes in the equations used to calculate ratings for each life stage, the decoupling of flow and temperature data from physical habitat survey data in the 3.1 version of HabRate strongly influenced the higher ratings, especially for summer rearing life stages of steelhead (0S and 1S) and chinook (0S). Accordingly, ratings generated using the 3.1 HabRate model differ from ratings generated under the 2.0 HabRate model for the exact same data, and re-combination of channel habitat units into new reaches may have further altered ratings by varying the habitat attribute values incorporated in the HabRate model for any given reach.

1997 Habitat Ratings

Habitat ratings for steelhead indicated that Whychus Creek provided adequate or optimal habitat for all steelhead life stages in 1997, with the exception of spawning and emergence (Table 2, Figure 1 to Figure 5). HabRate results suggest habitat in Whychus Creek was best for overwintering ages 0+ (0W) and 1+ (1W) steelhead. Habitat was adequate or optimal in all 16.3 miles surveyed in 1997 for summer rearing ages 0+ (0S) and 1+ (1S) steelhead.

Whychus habitat was least suitable for spawning and emergence life stages of steelhead in 1997. Reaches receiving poor ratings for spawning and emergence were limited by fine sediments exceeding the 20% maximum criterion for habitat to be suitable for spawning and emergence. Reaches providing suitable habitat for steelhead spawning and emergence were located between Camp Polk and the City of Sisters, and upstream of Sisters (Figure 1).

Table 2. 1997 Steelhead Habitat Ratings.

	Steelhead Life Stage									
1997 Reach Level Habitat Rating	Spawning and Emergence		Age 0+ Summer		Age 0+ Winter		Age 1+ Summer		Age 1+ Winter	
	Miles	%	Miles	%	Miles	%	Miles	%	Miles	%
Good	0.0	0%	3.7	22%	16.3	100%	0.0	0%	16.3	100%
Fair	2.7	16%	12.6	78%	0.0	0%	16.3	100%	0.0	0%
Poor	13.6	84%	0.0	0%	0.0	0%	0.0	0%	0.0	0%

All Whychus reaches surveyed in 1997 provided adequate or optimal habitat for both 0S and 0W Chinook rearing stages (Figures 2 and 3, Table 2). Habitat for Chinook spawning and emergence received a poor rating in the same 13.6 miles of creek receiving poor ratings for steelhead spawning and emergence, limited by fine sediments in excess of the same 20% maximum criterion. Habitat was adequate or optimal for Chinook spawning and emergence in 2.7 miles of creek between Camp Polk and the City of Sisters (Figure 1).

Table 3. 1997 Chinook Habitat Ratings.

1997 Reach Level Habitat Rating	Chinook Life Stage					
	Spawning and Emergence		Age 0+ Summer		Age 0+ Winter	
	Miles	%	Miles	%	Miles	%
Good	1.2	7%	0.5	3%	0.0	0%
Fair	1.5	9%	15.7	97%	16.3	100%
Poor	13.6	84%	0.0	0%	0.0	0%

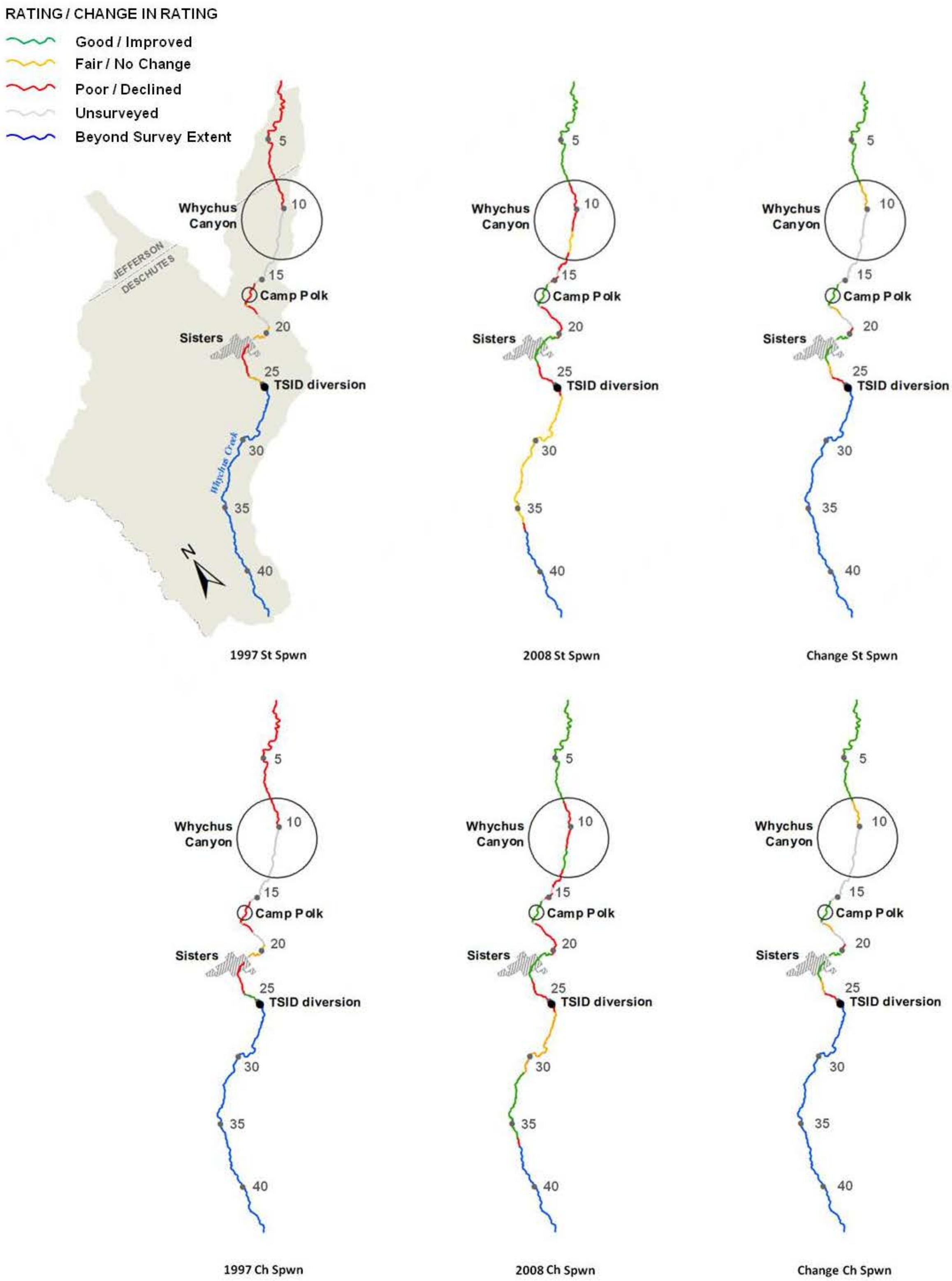

Figure 1. Stream reaches rated Good, Fair, and Poor in 1997 and 2008/2009, and changes in habitat quality ratings (Improved, No Change, or Declined) from 1997 to 2009, for spawning steelhead (Spwn)(upper panel) and chinook salmon (lower panel).

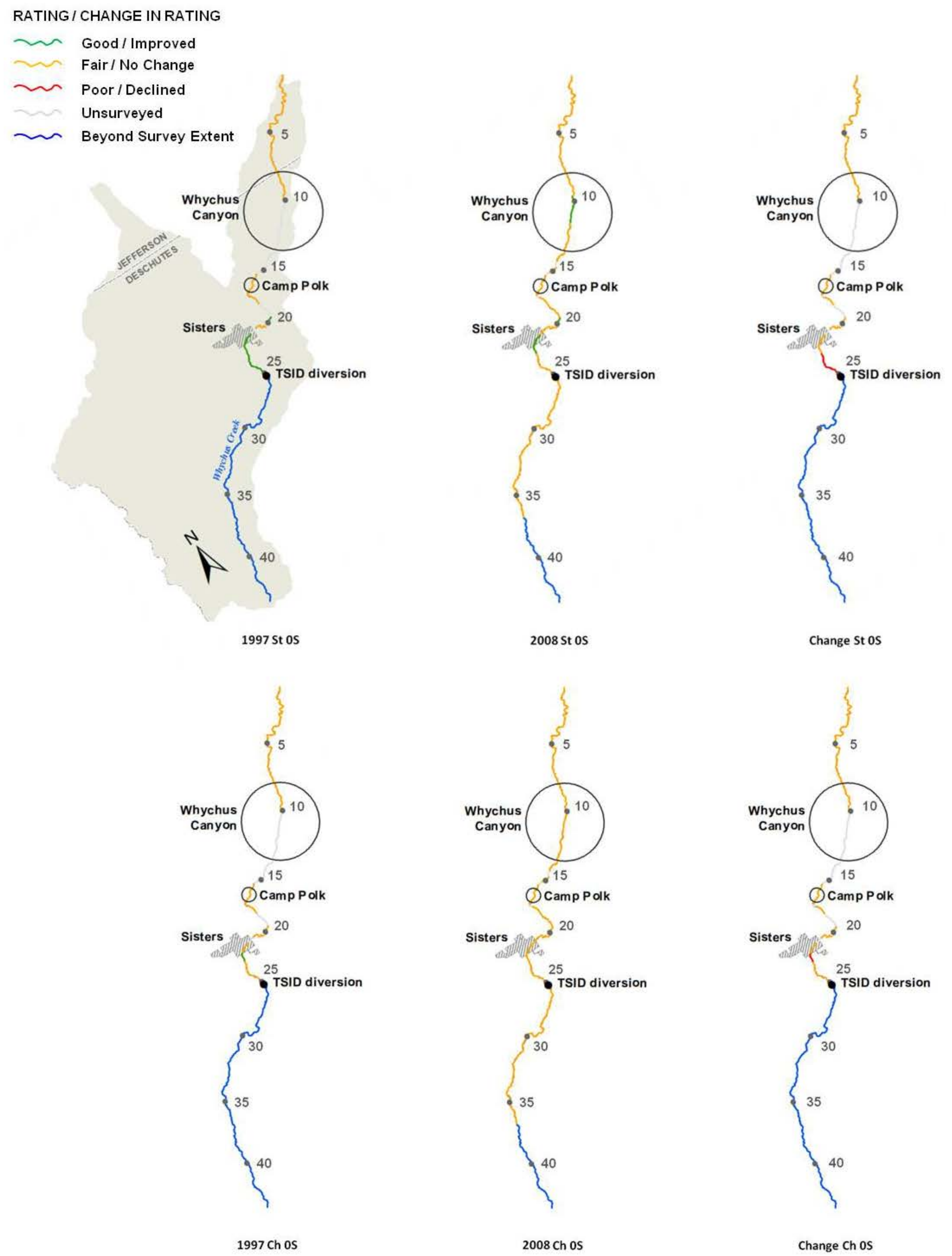

Figure 2. Stream reaches rated Good, Fair, and Poor in 1997 and 2008/2009, and changes in habitat quality ratings (Improved, No Change, or Declined) from 1997 to 2009, for age 0+ summer rearing steelhead (0S) (upper panel) and chinook salmon (lower panel).

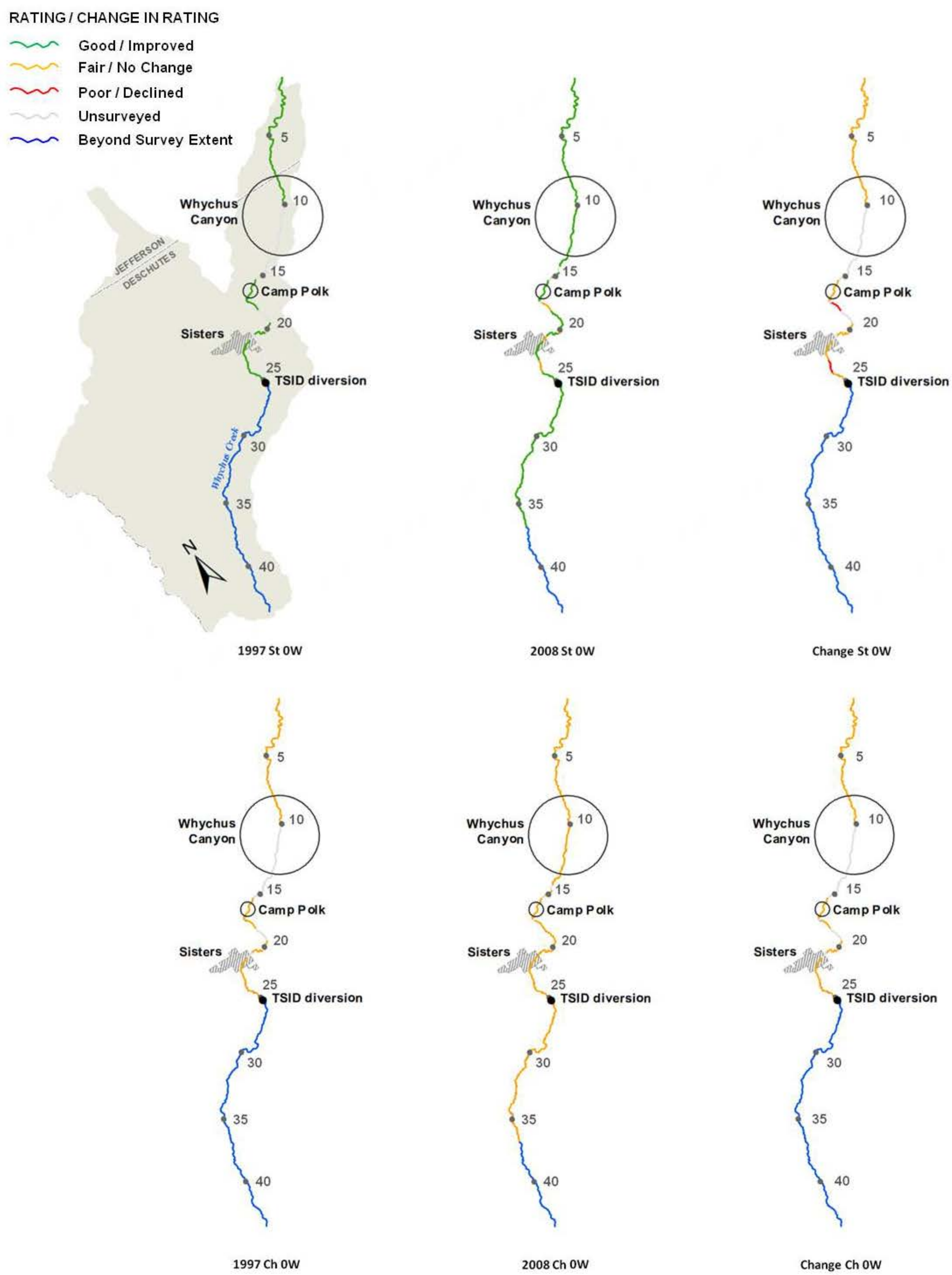

Figure 3. Stream reaches rated Good, Fair, and Poor in 1997 and 2008/2009, and changes in habitat quality ratings (Improved, No Change, or Declined) from 1997 to 2009, for age 0+ overwintering steelhead (0W)(upper panel) and chinook salmon (lower panel).

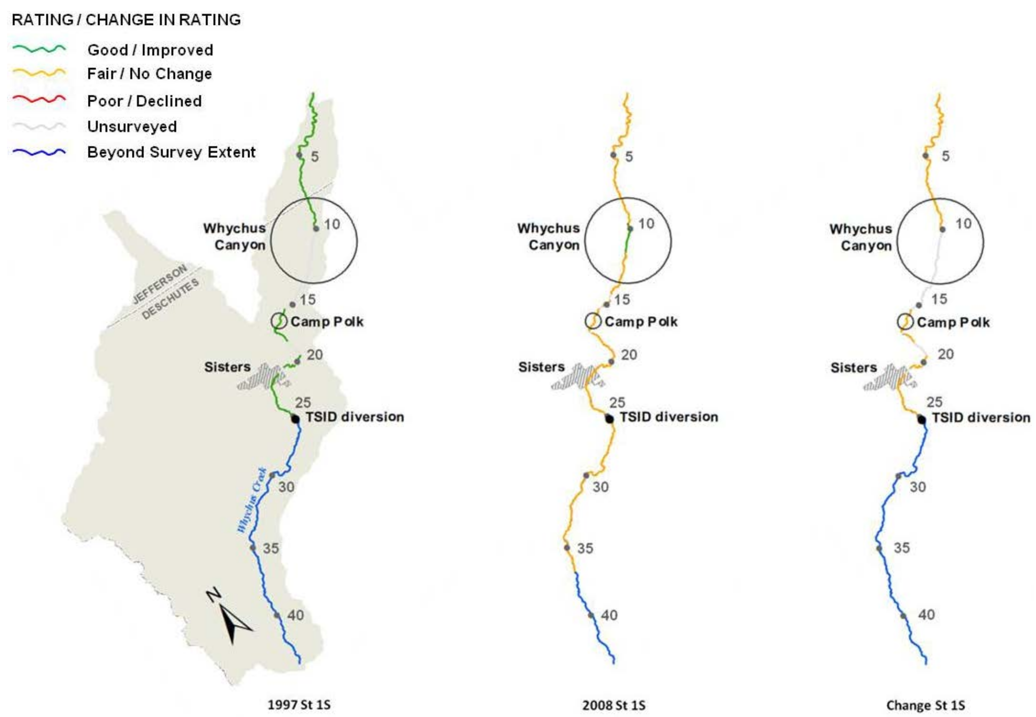

Figure 4. Stream reaches rated Good, Fair, and Poor in 1997 and 2008/2009, and changes in habitat quality ratings (Improved, No Change, or Declined) from 1997 to 2009, for age 1+ summer rearing steelhead (1S).

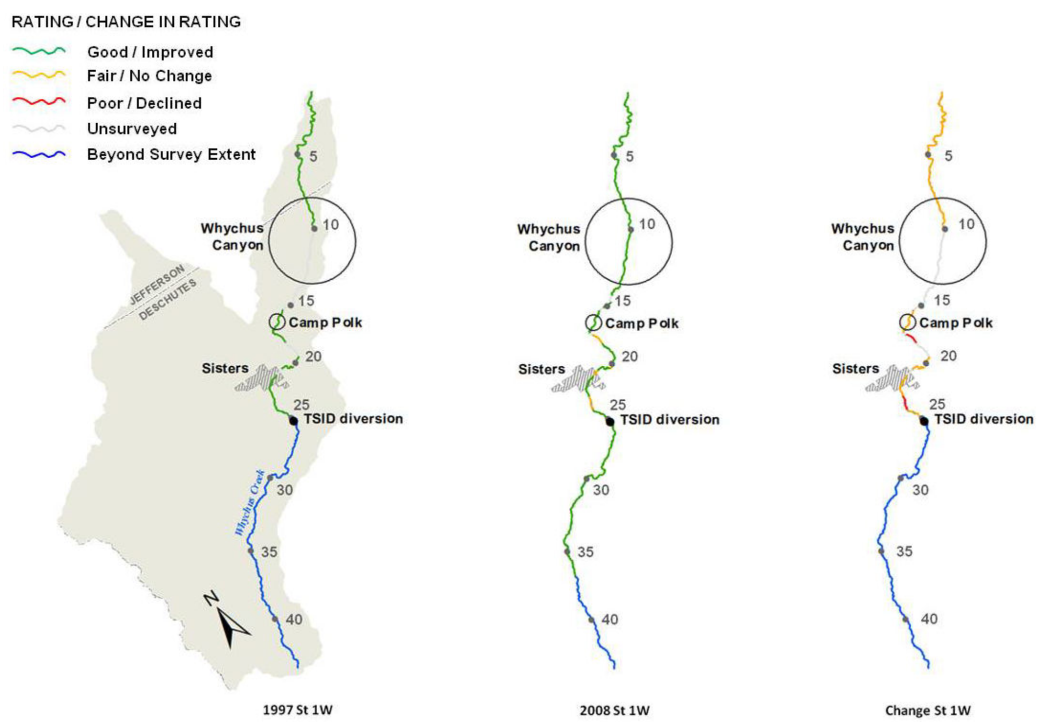

Figure 5. Stream reaches rated Good, Fair, and Poor in 1997 and 2008/2009, and changes in habitat quality ratings (Improved, No Change, or Declined) from 1997 to 2009, for age 1+ overwintering steelhead (1W).

2008/2009 Habitat Ratings

Trends for 2008-2009 habitat ratings for steelhead and chinook paralleled each other but differed substantially from 1997 ratings (Figures 1-5). All surveyed reaches were adequate or optimal for all summer rearing and overwintering life stages for both species (Table 4, Table 5). Spawning and emergence habitat was mixed for both species, with approximately a third of surveyed miles rated unsuitable for this life stage.

Habitat for steelhead spawning and emergence was adequate or optimal in 22.7 miles (67%) along Whychus in 2008/2009 (Table 4). The best spawning habitat occurred below river mile 8.0, along Camp Polk, and from just downstream to just upstream of the City of Sisters (Figure 1). Spawning habitat was adequate along a 1.3 mile reach between river miles 11.5 and 12.8 and from just above the Three Sisters Irrigation Diversion to just below the upper extent of habitat surveys near the headwaters of Whychus. Habitat was largely unsuitable for steelhead spawning and emergence between river mile 8.0 and Camp Polk, with additional unsuitable reaches between Camp Polk and the City of Sisters, and upstream of Sisters. Altogether 10.0 miles were unsuitable for steelhead spawning and emergence in 2008/2009. As with 1997 ratings, poor ratings for 8.2 of 2009/2009/2011 miles were driven by fine sediments in excess of 20%; the rating for the remaining 1.8 miles was a result of gravel percentages less than 15% and cobble less than 10%.

Habitat was most favorable for overwintering (0W and 1W) steelhead (Figures 3 and 5) (Table 4). Summer rearing (0S and 1S) habitat was adequate for all miles surveyed with the exception of a total of 3.4 miles rated “Good”, from river mile 9.7 to 11.4, a 0.4 mile reach downstream of Sisters, and along approximately a mile and a half approaching and above the upstream boundary of Sisters (Figures 2 and 4).

Table 4. 2008/2009 Steelhead Habitat Ratings.

2008 Reach Level Habitat Rating	Steelhead Life Stage									
	Spawning and Emergence		Age 0+ Summer		Age 0+ Winter		Age 1+ Summer		Age 1+ Winter	
	Miles	%	Miles	%	Miles	%	Miles	%	Miles	%
Good	11.9	36%	3.4	10%	31.0	93%	1.7	5%	31.0	93%
Fair	10.8	33%	29.8	90%	2.2	7%	31.5	95%	2.2	7%
Poor	10.5	32%	0.0	0%	0.0	0%	0.0	0%	0.0	0%

Spawning and emergence habitat was slightly better for chinook than for steelhead in 2008/2009/2011 (Table 5). Three reaches (5.8 miles) rated “Fair” for steelhead spawning and emergence were rated “Good” for chinook due to differences in optimal percentages of cobble for the two species; all other reaches received the same ratings for chinook spawning and emergence as for steelhead spawning and emergence (Figure 1). All surveyed reaches provided adequate habitat for summer (0S) and winter (0W) rearing chinook (Figures 2 and 3).

Table 5. 2008/2009 Chinook Habitat Ratings.

2008 Reach Level Habitat Rating	Chinook Life Stage: Spawning and Emergence		Age 0+ Summer		Age 0+ Winter	
	Miles	%	Miles	%	Miles	%
Good	17.7	53%	0.0	0%	0.0	0%
Fair	5.0	15%	33.2	100%	33.2	100%
Poor	10.5	32%	0.0	0%	0.0	0%

Changes in Habitat Ratings, 1997 to 2008/2009

Channel habitat unit data were available for comparison of change in habitat quality from 1997 to 2008/2009 for eleven reaches totaling 16.1 miles between river mile 0 at the mouth of Whychus Creek and river mile 25.2 at the TSID diversion. Ratings for spawning and emergence life stages of both steelhead and chinook improved from “Poor” to “Good” for five reaches totaling 10.6 miles, from the mouth of Whychus to river mile 8.0, through Camp Polk, and from the downstream boundary of Sisters to just upstream of Sisters (Table 6, Table 7). Spawning and emergence ratings improved from “Fair” to “Good” along one reach totaling 0.9 miles just downstream of Sisters. The improvement in spawning and emergence ratings in all reaches for both species, with the exception of Reach 16, immediately downstream of Sisters, for Chinook, resulted from reductions in fine sediments to less than 20%. The 1997 Chinook “Fair” rating for Reach 16 was not limited by fines but instead reflects missing data for the residual pool depth attribute. Spawning and emergence for both steelhead and Chinook declined along a total of 1.6 miles, in one reach immediately downstream of TSID and in a second reach downstream of Sisters (Figure 1). Declines in spawning and emergence habitat quality ratings for both species in these reaches resulted from an *increase* in fines to greater than 20%.

Habitat did not improve from 1997 to 2008/2009/2011 for any other steelhead or chinook life stage, and declined in some reaches for ages 0S, 0W and 1W steelhead and 0S Chinook (Figure 2, Figure 3, Figure 5) from optimal to adequate. Declines in habitat quality occurred for 0S steelhead in two reaches totaling 2 miles immediately downstream of TSID, driven by an overall decrease in cover characterized by undercut banks reduced to less than 10% and a reduction in the number of boulders to fewer than five per 100 m. Declines for 0W and 1W steelhead, in one reach upstream of Camp Polk and one reach upstream of Sisters totaling 1.6 miles, were also a result of decreases in boulders per 100 m and percent undercut banks. The decline in habitat quality for 0S Chinook in one half-mile reach immediately upstream of Sisters resulted from a decrease in percent pool area to less than 40%. There was no change in habitat quality from 1997 to 2008/2009 for 0W Chinook or for 1S steelhead (Figure 3, Figure 4).

Table 6. Changes in Reach Level Habitat Ratings for Steelhead Life Stages.

Change in Reach Level Habitat Rating	Steelhead Life Stage									
	Spawning and Emergence		Age 0+ Summer		Age 0+ Winter		Age 1+ Summer		Age 1+ Winter	
	Miles	%	Miles	%	Miles	%	Miles	%	Miles	%
Improved	11.4	71%	0.0	0%	0.0	0%	0.0	0%	0.0	0%
No Change	3.1	19%	14.1	88%	14.5	90%	16.1	100%	14.5	90%
Declined	1.57	10%	1.96	12%	1.56	10%	0.00	0%	1.56	10%

Table 7. Changes in Reach Level Habitat Ratings for Chinook Life Stages.

Change in Reach Level Habitat Rating	Chinook Life Stage					
	Spawning and Emergence		Age 0+ Summer		Age 0+ Winter	
	Miles	%	Miles	%	Miles	%
Improved	11.4	71%	0.0	0%	11.4	0%
No Change	3.1	19%	15.5	97%	3.1	100%
Declined	1.57	10%	0.54	3%	1.57	0%

Discussion

Results of the revised HabRate model suggest physical habitat conditions in 1997 were suitable for steelhead and chinook summer rearing and overwintering life stages in all reaches surveyed, but unsuitable for steelhead or chinook spawning in all but 2.7 miles surveyed. Of the 2.7 miles of habitat that were rated as suitable for steelhead and chinook spawning, 1.5 miles were located below the lowest confirmed barrier to fish passage (RM 22.2, Barrier No. 2, Leithauser diversion dam) and therefore likely would have been accessible to redband adults. No steelhead adults were present in Whychus in 1997.

Although ratings declined for some summer rearing and overwintering life stages of steelhead or chinook from 1997 to 2008/2009 from “Good” to “Fair”, the resulting 2008/2009 and 2011 ratings still indicated suitable habitat conditions for summer rearing and overwintering steelhead and chinook in all reaches surveyed in 2008/2009/2011. Of reaches surveyed in both 1997 and in 2008/2009, ratings indicated habitat was suitable (Fair or Good) for steelhead and chinook spawning and emergence in nine miles more in 2008/2009/2011 than in 1997, with a total of 22.7 miles of suitable spawning and emergence habitat for steelhead and chinook as of 2008/2009/2011 surveys. Almost thirteen miles of suitable spawning and emergence habitat were below the Leithauser diversion dam, the lowest passage barrier remaining as of 2008 (Mork 2012), and were therefore accessible to fish, while 9.4 miles were upstream of the Leithauser diversion dam at RM 22.2 and inaccessible for spawning.

The previous habitat quality report (Golden 2010) compared habitat quality ratings upstream of TSID in 1997 and 2008/2009. Because 1997 channel habitat unit data was unavailable for use in the updated version of HabRate, and because the updated model was different enough from older versions that results of the two versions are not comparable, we did not include 1997 data upstream from TSID in our analysis and thus make no conclusions about how habitat may have changed in these reaches.
Despite a net increase in ratings indicating suitable habitat for steelhead and Chinook freshwater life stages from 1997 to 2008/2009/2011, the changes observed in habitat ratings are not easily explained

by restoration actions that occurred between 1997 and 2011. Habitat quality ratings for Whychus Creek generated under HabRate 2.0 (Golden 2010) incorporated flow and temperature data and thus directly reflected differences recorded in flow and temperature habitat attributes resulting from increases in streamflow between 1997 and 2008/2009. But, because flow and temperature habitat attributes are not incorporated in HabRate 3.1, changes in habitat quality ratings presented in this analysis do not reflect changes in streamflow or associated changes in temperature between 1997 and 2008/2009. While the previous analysis (Golden 2010) would therefore have been expected to reflect changes in habitat quality that occurred as a result of increased streamflow during a period of extensive streamflow restoration, any such changes would not be reflected in HabRate 3.1 ratings. It is possible that changes in streamflow may have affected the habitat attributes driving differences in habitat quality ratings between 1997 and 2008/2009, and thus indirectly affected habitat quality ratings, but given the data available we are not able to account for any effects of streamflow on other habitat attributes. No channel restoration projects occurred in any reach surveyed between 1997 and 2011 that would explain the recorded differences in habitat attribute values and the consequent differences in habitat quality ratings.

One possible explanation for the differences in habitat quality ratings is sampling error between 1997 and 2008/2009 surveys. Differences in percentages of fines that determined changes in ratings between 1997 and 2008/2009 ranged between 4% and 11%; six fewer boulders in 2008/2009 than in 1997 resulted in the decline in ratings in one reach; percent undercut banks were less by 6% and 7% in two reaches in 2008/2009. The relatively small differences observed in estimates of fine sediments and percent undercut banks could easily have resulted from estimating differences between surveyors from one sampling period to the next, especially given the eleven to fourteen year interval between surveys.

Anecdotal and local knowledge of stream history and changes in stream conditions between 1997 and 2008/2009 suggest that changes in habitat attribute values and habitat quality ratings probably reflect differences in estimation of habitat attributes more than actual changes in those attributes. For example, a 7.8 mile reach of Whychus where fine sediments were reported to decrease from 1997 to 2008/2009 runs through a narrow canyon where the stream channel has been minimally modified by human activity, thus little habitat degradation or impairment of function due to human activity is likely to have occurred along the majority of this reach.

Given the lack of restoration projects that would be expected to result in a change in physical habitat conditions, and given anecdotal and local knowledge of stream reaches where habitat quality ratings changed from 1997 to 2008/2009, we are unable to attribute differences in habitat quality ratings to real changes in stream conditions. Accordingly, UDWC will use data and habitat quality ratings from 2008/2009/2011 as the baseline for monitoring changes in habitat conditions as channel restoration continues on Whychus Creek. UDWC will work with restoration partners to re-survey the same reaches used in this analysis as channel restoration projects are completed. Because channel restoration projects are anticipated to directly affect habitat attributes incorporated in HabRate, we expect habitat survey data and ratings generated from those data to reflect changes in habitat conditions resulting from channel restoration. Future analyses comparing habitat quality in restored reaches to 2008/2009/2011 conditions will help restoration partners understand how habitat conditions in Whychus Creek change following restoration activities.

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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 one mile to 11 miles. From 2009 to 2010 one barrier was retrofitted to provide fish passage, connecting two reaches and reducing total reaches to six. In 2011UDWC 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. UDWC reached an agreement with one water rights holder to remove a fourth barrier, and will continue to actively engage water rights holders to provide passage at the final two barriers by 2020. Removal of the three remaining barriers could 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 as described below, 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 2011.

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. From 2009 to 2010 and from 2010 to 2011 new fish passage projects were completed and others initiated. This report presents the status of fish passage and stream connectivity at the close of 2011.

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 2010 and from 2010 to 2011 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 14.7 to river mile 25.2 (Table 1, Figure 1). Barriers No. 1 and 2 were partial barriers, allowing anadromous fish at least intermittent access to a total of 21.3 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 two- and one-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 21.3, reducing the number of fragmented reaches to four and connecting two adjacent one-mile reaches to create a two-mile reach. During OWRD and UDWC inventories of existing fish passage barriers, surveyors were 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, six additional miles of habitat upstream of rm 14.7, and 20 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.

Over the course of 2011 UDWC and restoration partners continued to engage with water rights holders and landowners to plan fish passage restoration projects at remaining passage barriers. An agreement was reached and a design completed to remove one additional barrier, Barrier No. 2 at rm 20.9, in 2012. Restoration of fish passage at this point will increase total miles of habitat accessible from the mouth of Whychus Creek to 22 and reduce fragmented reaches below natural barriers to three.

Table 1.
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	14.7	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	20.9	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	21.3	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	22.3	44.2678	-121.5584	100%	4.5	48.0	18.0	Yes	Sokol irrigation diversion dam.	
No. 5	8/28/2009	23.6	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	25.1	44.2356	-121.5633	100%	3.2	45.0	43.0	Yes	McCallister irrigation diversion dam.	

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

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

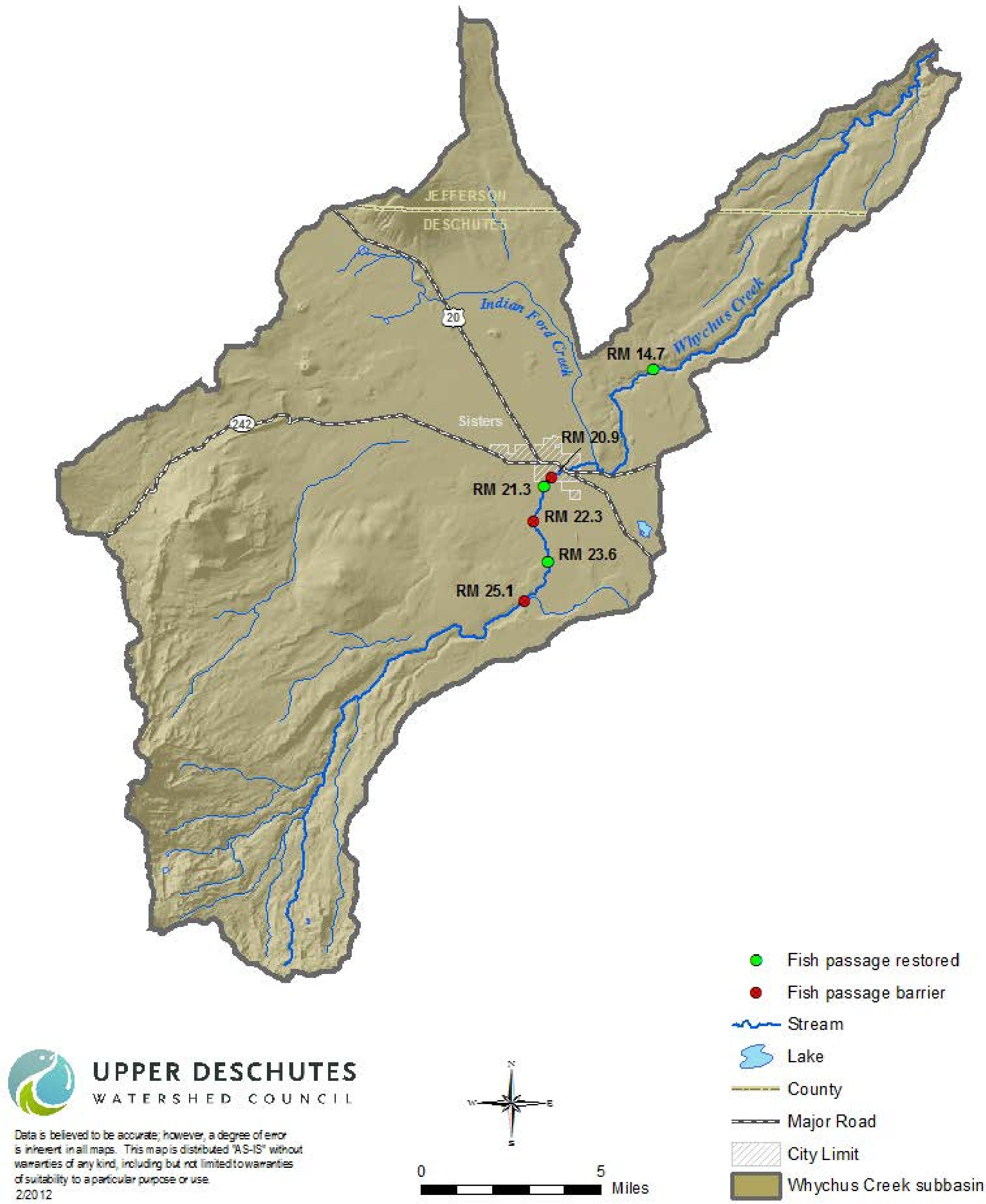

Figure 1.
In 2009, six barriers impaired stream connectivity on Whychus Creek between river miles 14.2 and 25.1. From 2009 to 2011 fish passage was restored or found to be intact at three of these barriers. UDWC and restoration partners aim to provide passage at each of these barriers by 2020.

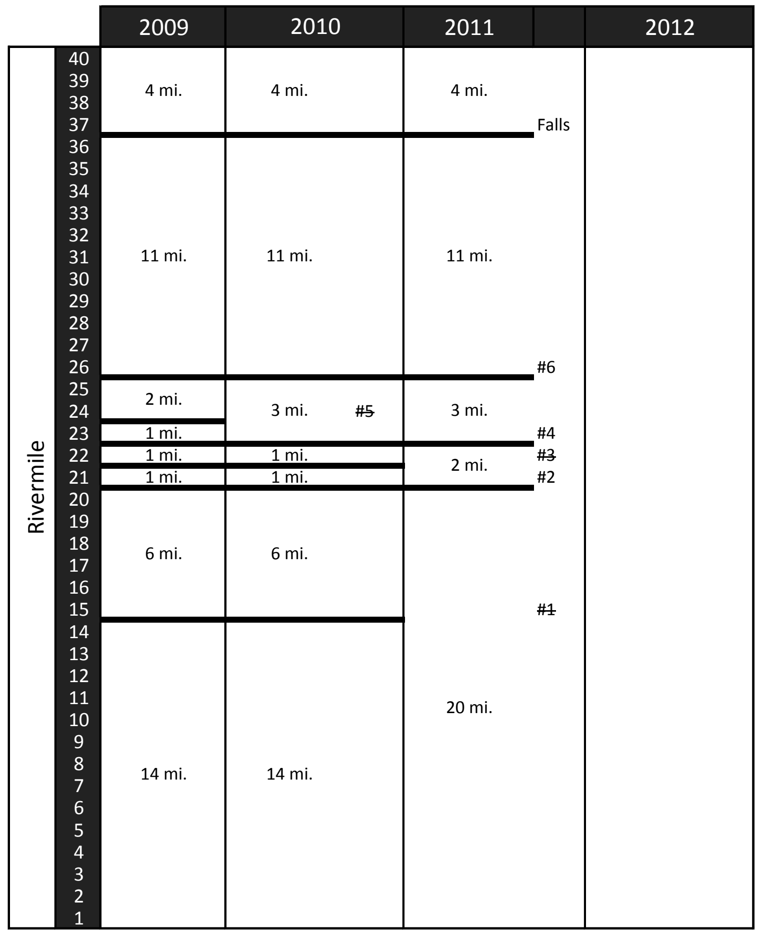

Figure 2.

In 2011 fish passage was restored at Barrier No. 3, and Barrier No. 1 was determined not to restrict fish passage, reducing the number of barriers limiting connectivity in Whychus Creek to four (numbered per Table 1), and reducing the number of fragmented reaches to four varying in length from two miles to 20 miles. Whychus Creek Falls, located between river miles 36 and 37, is a natural barrier.

Discussion

Existing barriers determine the number of miles of contiguous stream habitat accessible to fish. Over time, as barriers are removed, habitat connectivity will increase. 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 could 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. As these projects are implemented and this inventory updated every year, it will be possible to monitor the additional river miles of habitat opened to anadromous and resident fish.

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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 on 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 81% reduction from 2009 baseline unscreened flows. As of 2011, 37.77 of the total 178.02 cfs diverted for irrigation remain unscreened. 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 2011.

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 on 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.

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. New fish screens are built to specifications that meet ODFW and NMFS fish screening criteria.

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 criteria of fish screening for both anadromous and native resident fish species. As one mode of establishing a baseline of risk factors linked to fish entrainment for future years, 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 a consequent reduction in the potential for fish entrainment.

Results

The 2009 baseline inventory identified 13 active irrigation diversions extending from river mile 9.25 to river mile 25.25, 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, 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. Two additional diversions (Diversions No. 5 and 7) were screened to meet NMFS and ODFW criteria. Flows associated with these screens total 2.12 cfs, reducing the cumulative unscreened diversion rate to 175.45 cfs, 98.5% of the 2010 total diversion rate. In April 2011, construction of the fish screen at the Three Sisters Irrigation District diversion was completed. At 137.68 cfs, the TSID diversion represents the single largest flow withdrawal on Whychus Creek. Completion of the TSID fish screen reduced the cumulative unscreened diversion rate on Whychus Creek to 37.77 cfs representing just 21% of the 2010 total diversion rate. UDWC also reached agreements with water rights holders in 2011 to decommission and remove two additional diversions, No. 3 (rm 23.9) and No. 7 (rm 20.9), eliminating the risk of fish entrainment at these locations. Removal of these diversions will reduce cumulative unscreened flows by 6.64 cfs, to a new low unscreened rate of 31.13 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).

UDWC and ODFW, along with many of their partners, continue to actively work with landowners and water right holders to reach agreements to screen three of the remaining eight irrigation diversions to meet state and federal criteria and reduce the risk of entrainment for both anadromous and native fish species.

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. Three of the original twelve unscreened diversions were screened from 2009-2011, leaving nine diversions unscreened. Diversion No. 11, the Meyer diversion, has been determined not to pose an entrainment risk to fish.

2002-2009 Baseline data								2009 - 2011			
Diversion ID	Baseline Sampling Date	River Mile	Diversion Type	Associated Diversion Rate (cfs)	Screen Present at Baseline Inventory	Screen opening size (inches)	Met State & Federal Criteria at Baseline Inventory	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	5.52	May-12	No	Lazy Z / Uncle John
No. 4	8/28/2009	23.65	Gravity	153.00	No	N/A	No	137.68	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	1.12	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 **2011 Diversion Total 178.02**

Baseline Unscreened Total 192.89 **2011 Unscreened Total 37.77**

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

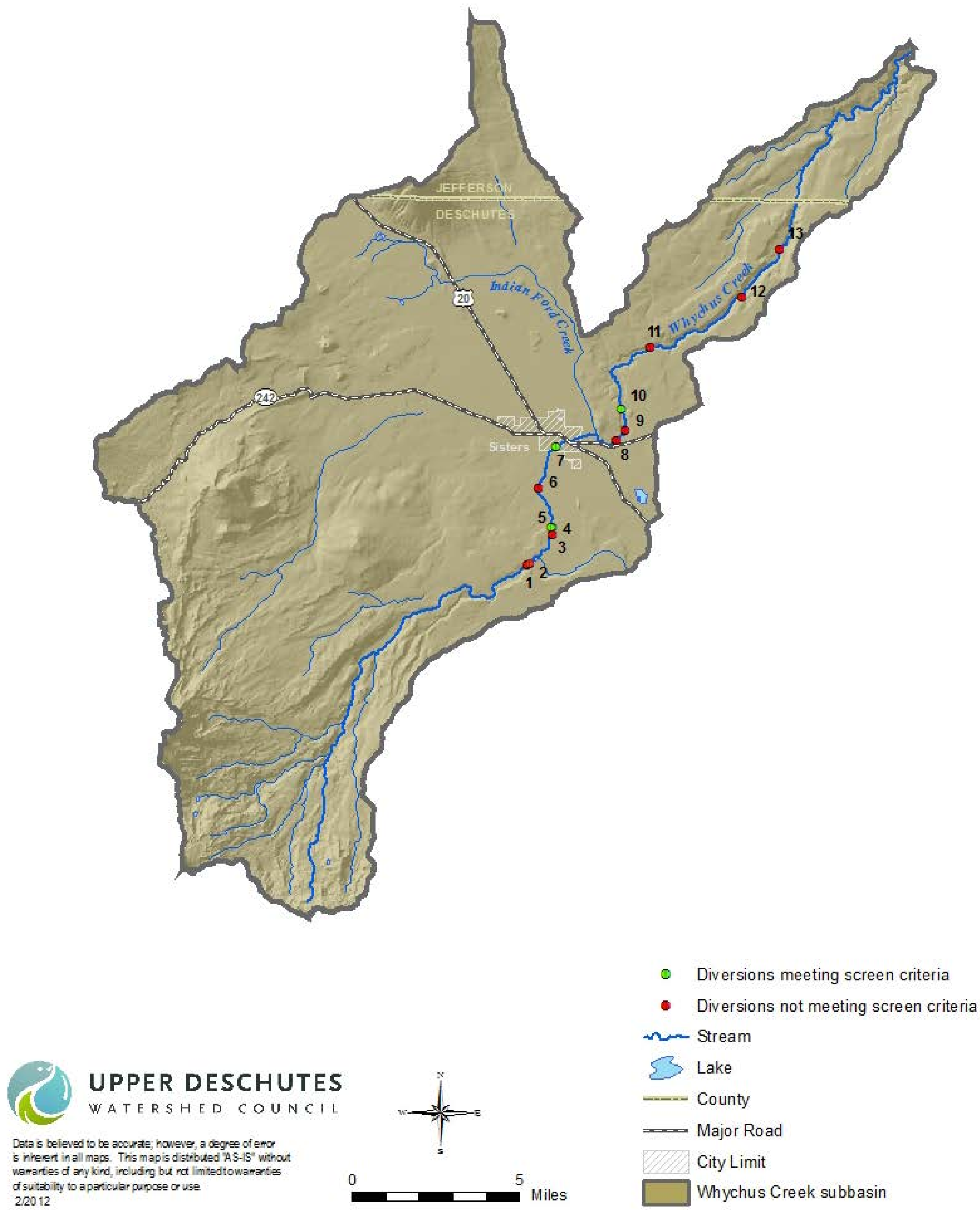

Figure 1.
In 2009 UDWC identified 13 irrigation diversions including 12 that did not meet federal and state criteria for fish screens on Whychus Creek (NMFS 2008). One diversion identified during baseline data collection met ODFW and NMFS screening criteria. From 2009 to 2011 two diversions were screened to reduce cumulative unscreened flows to 37.77. 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 2011, the cumulative unscreened diversion rate had been reduced by 81%, from 192.89 to 37.77 cfs. Screening projects planned for 2012 will further reduce unscreened diverted flows to 31.13 cfs. 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, and 2011

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Abstract

Aquatic macroinvertebrate monitoring was conducted in 2005, 2009, and 2011 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 activities on stream biota. On each occasion, Xerces staff trained volunteers in standardized stream sampling techniques; volunteer teams then dispersed to sample reaches at pre-determined sites. The high level of similarity among replicate sites sampled for quality assurance in each year indicates that trained volunteers implemented the protocol successfully, and that differences between sites and years are unlikely to be the result of operator error. The benthic macroinvertebrate community in Whychus Creek is responding to habitat restoration; the overall composition of the community changed substantially from 2005 to 2009, but remained more stable from 2009 to 2011. Biotic conditions reflected by PREDATOR and IBI scores show sustained improvement among sites in downstream reaches of the creek, with less improvement in the mid-stream reaches and a downward trend in biological condition among upstream sites by 2011. In contrast, the proportion of all collected taxa comprised of sensitive EPT (Ephemeroptera, Plecoptera, Trichoptera) as well as the relative abundance of EPT individuals was significantly higher among all upstream sites in 2011 compared to 2005. Community data from 2011 may be anomalous due to the effects of an unusually snowy winter and cold wet spring, leading to higher faster flows and colder temperatures throughout the season. However, as these data suggest decreased abundance and lower biotic conditions among upstream sampling sites, land use in this area should be examined for any changes from 2009 that might account for diminished stream health. Annual sampling at these sites, ideally for several years following planned channel reconstruction at Camp Polk, will contribute to effectiveness monitoring and help reveal the response of the benthic macroinvertebrate community to existing and continuing restoration projects.

Project Background

Biomonitoring in Whychus Creek

Biomonitoring evaluates the biological health of a body of water by examining the state of its biotic communities, such as plants, amphibians, algae, diatoms, or invertebrates (Rosenberg & Resh 1993, Karr & Chu 1999). If the habitat is impaired, the structure of these communities will be altered, according to individual species' sensitivity or tolerance to different stressors. Benthic macroinvertebrates are ideal tools for biomonitoring because:

- They process nutrients and energy, and are a critical part of the food web. Restoration work targeted at native fish, for example, is unlikely to be successful in the absence of an aquatic invertebrate food base, as the quantity and quality of prey items can limit the growth and survival of juvenile salmonids (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 or, in the case of more mobile winged adult taxa, migrate out of an area.
- They exhibit a range of responses to human-induced stressors, such that changes in different groups may reflect the effects of temperature, sediment, or flow.
- They have a short generation time that allows changes in their community structure to be detected rapidly following a disturbance.
- They are ubiquitous and 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, and 2011 from sites spanning RM 30.25 to RM 0.5. The sampling in 2005 occurred prior to any 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 conducted, 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. It is anticipated that monitoring will be done in the future to continue to assess changes in invertebrate community structure resulting from completed projects as well as from returning the creek to its historic meandering channel at Camp Polk.

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 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 sites. 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.

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.). Thus, stream condition rankings based on IBI scores may be less accurate in different parts of the state.

Methods

Sampling Sites

Ten sites along Whychus Creek were sampled in 2005 and 2009; 13 sites were sampled in 2011. 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. Overall, sampling sites are distributed broadly along the stream into downstream (RM 0.5- 9.5), mid-stream (RM 18-19.5), and upstream reaches (RM 23.5 - 30.25). A map of the sampling sites is shown in Appendix A.

Volunteer Training & Sampling Techniques

Training

On August 20 in 2005 and 2011 and on August 21 in 2009, Upper Deschutes Watershed Council staff and volunteers assembled at City Park in Sisters, OR and were trained by Xerces staff in macroinvertebrate monitoring protocols established for Oregon's wadeable streams (OWEB, 2003). Sampling protocol was demonstrated and each item on the datasheet was explained (see Appendix B for data sheet). Additional handouts on macroinvertebrate identification, including field guides to Northwest stream macroinvertebrates (Adams *et al*. 2003) and freshwater mussels (Nedeau *et al*. 2009) were provided, although volunteers were not expected to identify any organisms collected. The group was 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, 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

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
WC00300[a]	RM 3, u/s Alder Springs	44.43458, -121.35976	2005
WC00600[b]	RM 6, u/s Rd 6360	44.40412, -121.40259	2005, 2009, 2011
WC00875	RM 8.75, Rimrock Ranch d/s	44.391278, -121.406182	2011
WC00900	RM 9, Rimrock Ranch	44.384198, -121.407892	2005, 2009, 2011
WC00950	RM 9.5, Rimrock Ranch u/s	44.371534, -121.415865	2011
WC01800	RM 18	44.328342, -121.494534	2005
WC01825	RM 18.25, d/s end DBLT property	44.32781, -121.495406	2009, 2011
WC01850	RM 18.5, DBLT property	44.326601, -121.500229	2009, 2011
WC01900	RM 19, DBLT property	44.321523, -121.507461	2005, 2009, 2011
WC01950	RM 19.5, d/s Camp Polk Bridge on DBLT	44.318741, -121.514961	2009, 2011
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
WC02600[c]	RM 26, 4606 Rd. footbridge	44.2730592, -121.555297	2005, 2009, 2011
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

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 the sampling reach themselves; in 2011, this process was greatly streamlined, as watershed council staff calculated wetted widths and flagged the upstream and downstream extent of each sampling reach one day 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. 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 bucket and large debris was rinsed and removed, as were any fish or amphibians collected incidentally. Sample material was then poured through a sieve to remove the water, and the composited material was placed into 1-liter Nalgene jars with 80% ethanol added as a preservative. In cases where excessive amounts of sand and gravel were collected, the sample was elutriated by adding water to the sample bucket, swirling it 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.

Jars were filled no more than halfway with sample material to ensure adequate 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).

Identification

Samples collected in 2005 were identified by Aquatic Biology Associates, Inc. (ABA; Corvallis, OR); those collected in 2009 and 2011 were identified by ABR, Inc. Environmental Research & Services (ABR; Forest Grove, OR). Each composite sample was randomly sub-sampled to a target of 500 organisms. In 2005, this 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.

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 expected to occur at each site but absent (missing taxa) as well as observed taxa that were 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 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.

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 conducted using the PRIMER V6 ecological community statistics software package (Clarke & Warwick 2001). CLUSTER analysis was conducted in PRIMER on a Bray-Curtis similarity matrix of square-root transformed data to investigate macroinvertebrate community similarity between sites and across years.

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.

Site O/E

PREDATOR scores showed a general improvement from 2005 to 2009; however, in 2011, most sites either had a similar or lower biological condition compared to 2009 (Figure 1; and see Appendix C for individual PREDATOR site scores). 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 two sites as most disturbed, five sites as moderately disturbed, and three as least disturbed. It is worth noting that the two most disturbed sites in 2009 were located near the headwaters of the creek (RM 26 and RM 30.25), thus this scoring may have been influenced by the fact that the glacier-fed, high-elevation headwaters of this stream may support a more limited macroinvertebrate community. Ten of the 13 sites sampled in 2011 were also sampled in 2009. Five of those 10 sites had PREDATOR scores that reflected the same biological condition as in 2009, with RM 1.5, 6.0, and 9.0 scoring as moderately disturbed in both 2009 and 2011, and RM 26.0 and 30.25 scoring as most disturbed in both years. PREDATOR scores at the remaining five sites were uniformly lower, such that sites that scored as either least or moderately disturbed in 2009 all scored as most disturbed in 2011.

In comparing PREDATOR scores across all three sampling years, the downstream reaches (RM 0.5 to 9.5) showed the most sustained improvement, with a mean PREDATOR score of 0.68 in 2005 increasing to 0.86 and 0.82 in 2009 and 2011, respectively. The midstream reaches (RM 18-19.5) remained in roughly the same condition from 2005 to 2009, with all sites scoring as least disturbed in both years, with the exception of RM 18.5, which scored as moderately disturbed (though close to the border between moderately and least disturbed). The mean PREDATOR score across these reaches was similar in both 2005 and 2009 (1.16 and 0.98, respectively). However, all of these midstream sites scored substantially worse in 2011, with PREDATOR scores reflecting most disturbed conditions, and a mean PREDATOR score across these reaches of only 0.65. A similar situation was seen for the upstream reaches (RM23.5 to 30.25), with almost identical mean PREDATOR scores across these reaches from 2005 to 2009 (0.76 and 0.75, respectively) dropping to a mean of 0.64 in 2011.

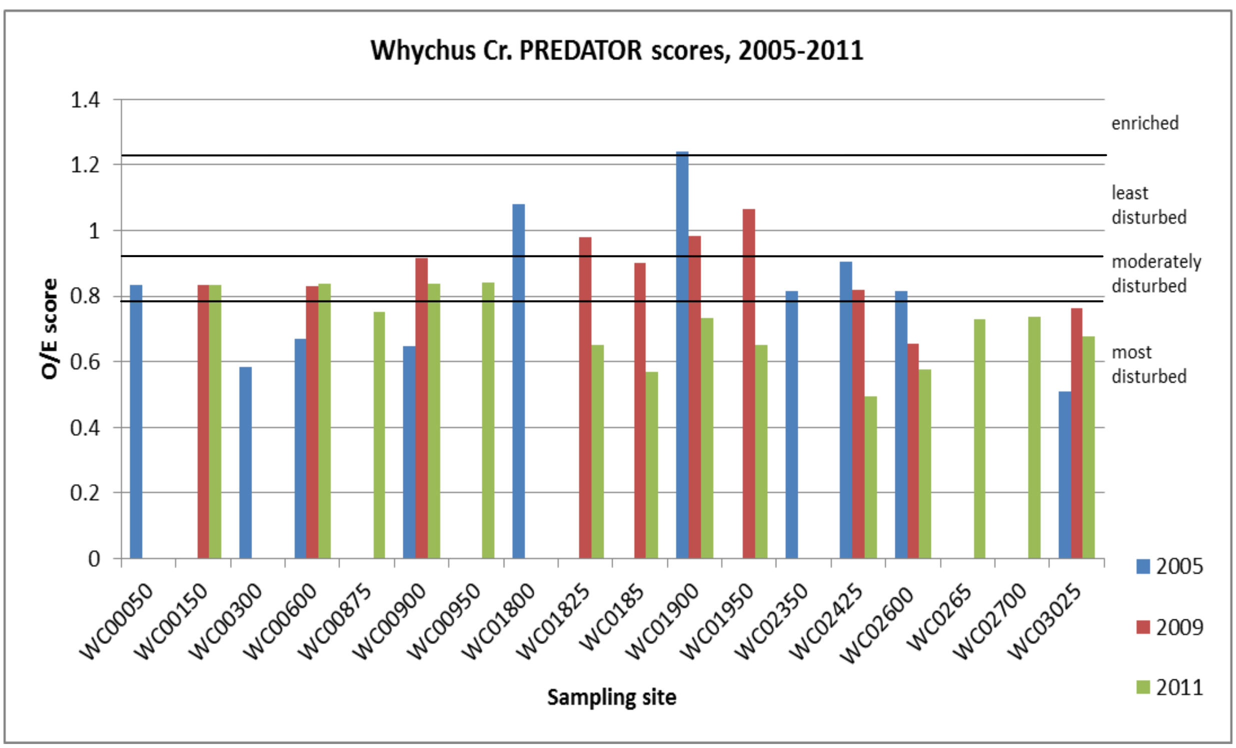


Figure 1. Whychus Creek sample site PREDATOR scores (Western Cordillera + Columbia Plateau, p_c>0.5)

The question is whether the decreases in PREDATOR scores observed in 2011 in the middle and upstream reaches of the creek reflect a true decrease in biological quality since 2009 due to anthropogenic stressors, or if other factors are influencing the macroinvertebrate community. As noted above, macroinvertebrate abundance was also lower in 2011, especially in the upstream reaches, with only 7 of the 13 sampling sites yielding the desired target number of 500 organisms. An aspect of the conditions in 2011 that should be considered is the effect on this glacier-fed stream of an extremely snowy winter in 2010/2011 followed by a long, wet, cold spring. Water temperatures were lower by 1.5 to 5.5 °C at six of the sites sampled in 2011 compared to 2009, and the mean temperature across all sites was also lower in 2011 (13.9°C) than in 2009 (15.4°C). The late-summer water level and flow rate also seemed higher in 2011, such that some of the sampling sites were almost too deep and fast to allow use of the kick net. Thus, although conditions at these sites should be investigated for any changes in human impacts, it is also possible that lower PREDATOR scores in 2011 may have been caused by faster flows and lower temperatures throughout the season that scoured out some groups of macroinvertebrates and rendered conditions unsuitable for others.

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 both missing and replacement taxa from 2005 to 2011. In 2005 and 2009, taxa that were absent from 7 or more of the 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 ≥7 of the sites where they were expected to occur. Taxa missing from ≥7 expected sampling sites in 2011 included many of the above (Leptophlebiidae, *Malenka*, Pisidiidae, Tanypodinae, and *Calineuria*), as well as Chironomidae (a common non-biting midge group), *Optioservus* and *Zaitzevia* (tolerant riffle beetle genera), and *Hydropsyche* (a tolerant net-spinning caddisfly).

Substantial similarity was also seen among replacement taxa found at ≥7 sampling sites across the three years of sampling, with 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) present as replacement taxa in all years. Additional replacement taxa seen in 2011 included Nematoda (common roundworms), Turbellaria (flatworms), *Drunella* (a sensitive spiny crawler mayfly genus), *Rickera* (a sensitive stripetail stonefly genus), and *Brachycentrus* (a sensitive genus of humpless case-making caddisfly).

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 both 2005 and 2011, the mean sediment optima for replacement taxa was significantly lower than that of the missing taxa (P= 0.0381 and 0.0035, respectively); in 2009, the mean sediment optima was also substantially lower among replacement taxa, although the difference was not quite significant (P =

0.0559). It is interesting that lower sediment optima was seen among replacement taxa in 2011, when higher precipitation and faster flow rates may have been expected to increase substrate mobilization. The consistent lower sediment optima among replacement taxa across all years of sampling may reflect overall changed conditions following restoration activities.

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 (P = 0.0481). Stream conditions in 2011 differed from those seen in earlier years, due to an extremely snowy winter in 2010/2011 followed by a long wet spring, and higher water levels and colder temperatures could account for some of the differences seen in temperature optima in that sampling year alone. However, restoration of instream flow accomplished in Whychus Creek may be sustaining lower temperatures compared to past conditions. Continued sampling and examination of seasonal water temperature data will help determine if this is a real trend as opposed to a single-year outlier.

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, and the corresponding scaled IBI score is in parentheses:

- 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 and no sites were scored as severely impaired in any year (Figure 2; see Appendix C for individual site IBI scores).

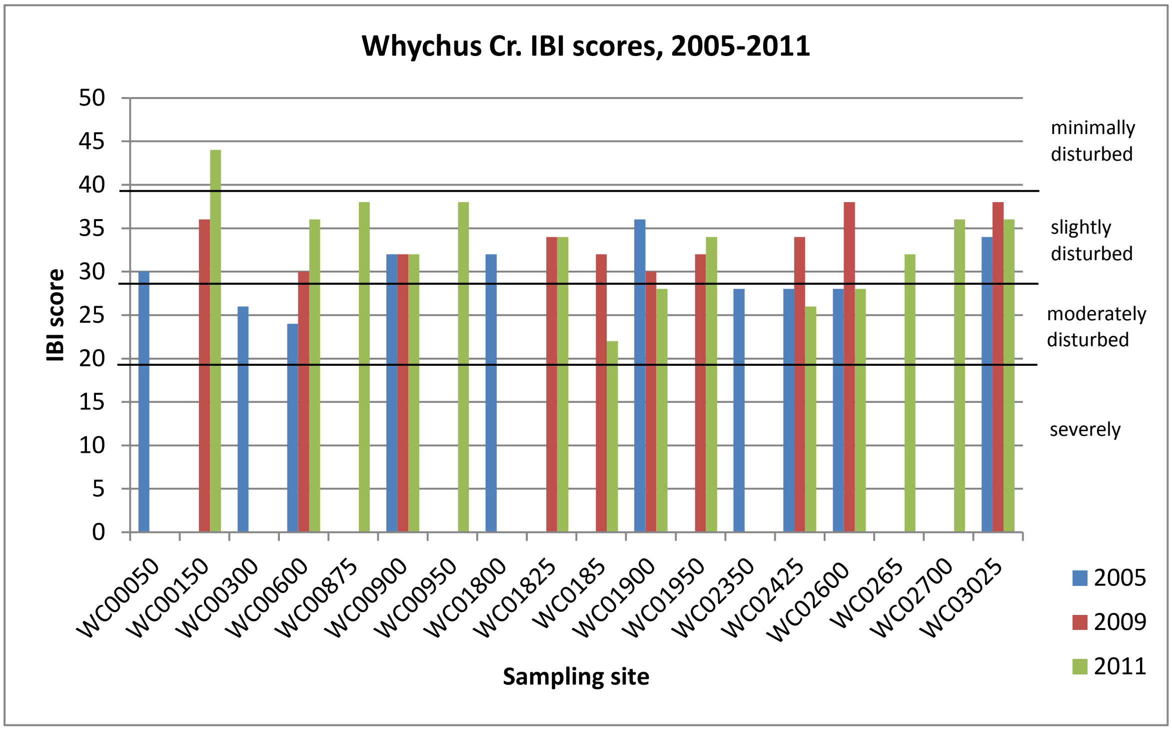


Figure 2. Level 3 IBI scores for Whychus Creek sites

As was seen with PREDATOR scores, IBI scores across all years reflected a more sustained improvement in biological condition for sites in downstream reaches, with mean IBI scores for sampling sites within RM 0.5 to 9.5 increasing from 28 in 2005 to 32.7 and 37.6 in 2009 and 2011, respectively. Mid-stream reaches had similar IBI scores in 2005 and 2009 (34 and 32, respectively), decreasing slightly to 29.5 in 2011. Mean IBI scores for the upstream sampling sites (RM 23.5 to 30.25) showed a dramatic increase from 29.5 in 2005 to 36.7 in 2009, and then decreased to 31.6 in 2011.

To assess whether particular metrics accounted for the majority of change noted in the scores from year to year, the values of each of the 10 metrics were examined for each site sampled in either two or three consecutive years (see Appendix D). While the values for most metrics changed to some extent from year to year at a site, the greatest variation was seen for % dominant top taxon, % tolerant taxa, and % sediment tolerant taxa, again implicating sediment as a possible driver for community changes.

Macroinvertebrate Community Composition

EPT

The three most sensitive groups of aquatic macroinvertebrates, i.e. mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera), referred to collectively as EPT, are often used as a measure of biological quality. Most taxa in these families require cold, clean, well-oxygenated water, and a greater abundance and diversity of EPT taxa is considered to correlate with better stream conditions. Although PREDATOR and IBI scores for each site changed from year to year, with some downward trend observed in 2011, the total number of EPT taxa 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

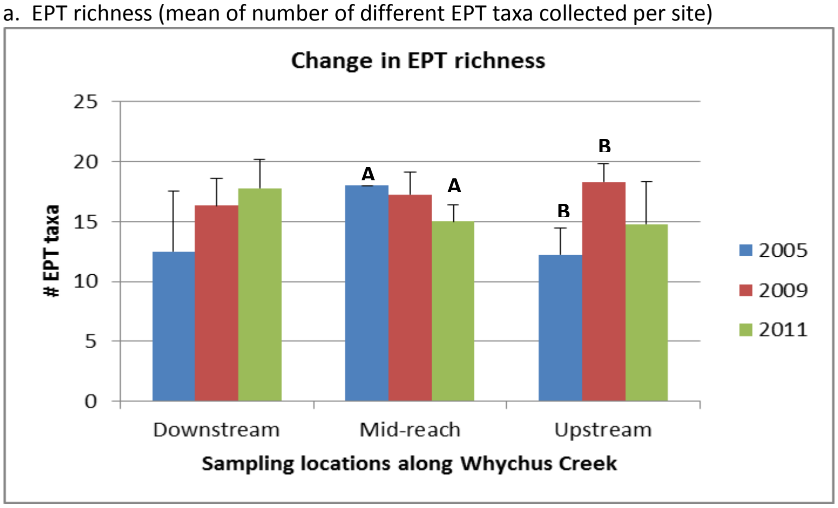
Trichoptera). In 2011, 82 taxa were collected across all sampling sites, including 49 EPT taxa (17 Ephemeroptera, 13 Plecoptera, and 19 Trichoptera). The proportion of total taxa comprised of EPT was 55% in both 2005 and 2009, and slightly higher in 2011 (59.8%).

Because macroinvertebrates respond to aspects of microhabitat that differ among stream reaches, changes in EPT composition across sampling years were also 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 3). In the downstream reaches, which had also showed a tendency towards improving biological conditions based on PREDATOR and IBI scores, there was 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 compared to 2009 (P=0.0397; Figure 3C). The mean number and proportion of taxa and relative abundance of EPT was similar among all mid-reach sites from 2005 to 2011; the number of EPT taxa collected was significantly lower in 2011 compared to 2005 (P=0.047; Figure 3A), though this represented a difference of only 3 taxa.

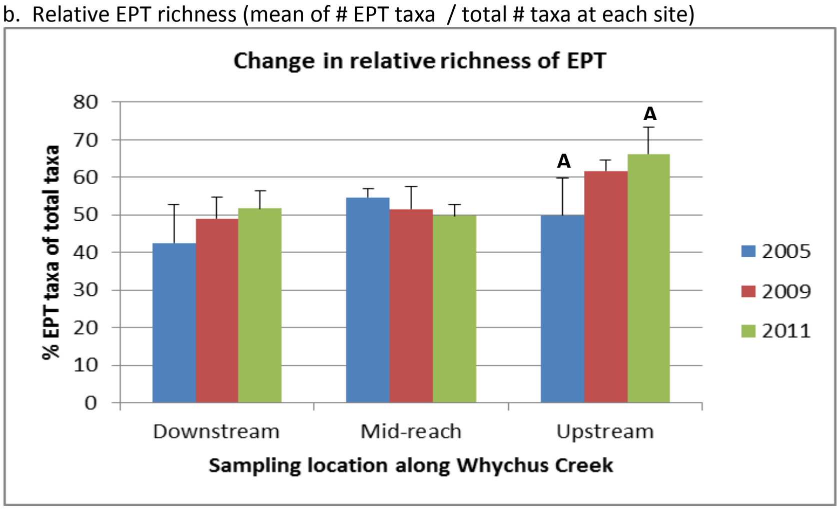
The most significant change in EPT composition was seen among upstream sampling sites; paradoxically, although these sites showed a downward trend based on PREDATOR and IBI scores in 2011, EPT diversity and abundance trended upwards, and both the proportion of total taxa comprised of EPT as well as the relative abundance of EPT at each site were significantly greater in 2011 compared to 2005 (P = 0.0221 and 0.0006, respectively; Figure 3B, C). Most EPT taxa are adapted to live in cold, fast-flowing waters, so their increased abundance in the upstream sampling sites in 2011 may be a reflection of the colder wetter conditions experienced the winter and spring prior to sampling.

Figure 3. Changes in EPT taxa. Letters indicate significant difference between mean values (P<0.05)

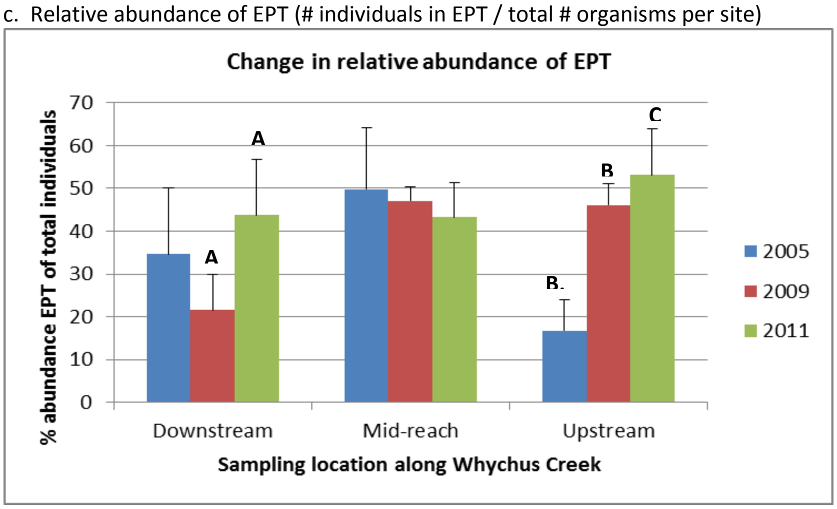
a. EPT richness (mean of number of different EPT taxa collected per site)



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

Many taxa were found at only a single site (29 taxa in 2005, 30 taxa in 2009) 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; these taxa were also present at very low abundance. Of the 118 total taxa collected during this project, 33 were present in all three 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 few instances where a difference was seen in the presence or absence of a certain group in 2005 compared to later sampling years, the most notable being *Rhyacophila* caddisflies, which were absent from all samples in 2005 but present as multiple different species in both 2009 and 2011, especially in the mid- to upstream reaches of the creek. In addition, 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. Rhyacophilids are predators that prefer cool flowing water, while limnephilids are shredders that can often be found in more lentic and/or temporary waters; changes in stream flow and temperatures following restoration may have influenced this shift.

Analysis of a Bray-Curtis similarity matrix of square root-transformed data suggests an overall change in community composition occurred from 2005 compared to 2009 and 2011. Mean macroinvertebrate community similarity among the five sites that were sampled in all three years (WC00600, WC00900, WC02425, WC02600, and WC03025) was lowest for 2005 compared to 2011 (mean similarity among all sites = 31.7) and highest for the same sites in 2009 compared to 2011 (mean similarity among all sites = 48.6). The four sites sampled only in 2009 and 2011 (WC01500, WC01825, WC01850, and WC01950) had the highest mean Bray-Curtis similarity, at 55.4, again suggesting a smaller degree of change in community composition in the later sampling years.

The idea that an overall shift in macroinvertebrate community composition occurred between 2005 and 2009-2011 is further supported by CLUSTER analysis (PRIMER V6), which grouped all 2005 samples separately from all 2009 and 2011 samples, with an average similarity between the 2005 sample cluster and the 2009+2011 cluster of only 29% (Figure 4). Clustering of the 2009 and 2011 samples was 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. For example, samples from RM 1.5, 6, and 9 taken in 2009 clustered most closely with samples taken from the same location and additional locations within the same area of the stream in 2011 (RM 1.5, 6, 9, 8.75, and 9.5), with an average similarity of 52%. Similarly, mid-reach samples taken in 2009 (RM 18.25, 18.5, 19, and 19.5) clustered most closely with samples taken at the same four sites in 2011, with an average similarity of 54%; and samples taken in 2009 at upstream portions of the creek (RM 24.25, 26, and 30.25) clustered with an average similarity of 43.5% with all upstream samples taken in 2011.

Replicate samples (DUP) taken in all three years for quality assurance purposes clustered closely with each other in each 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 in macroinvertebrate community composition between sites and years are real and not an artifact of operator errors.

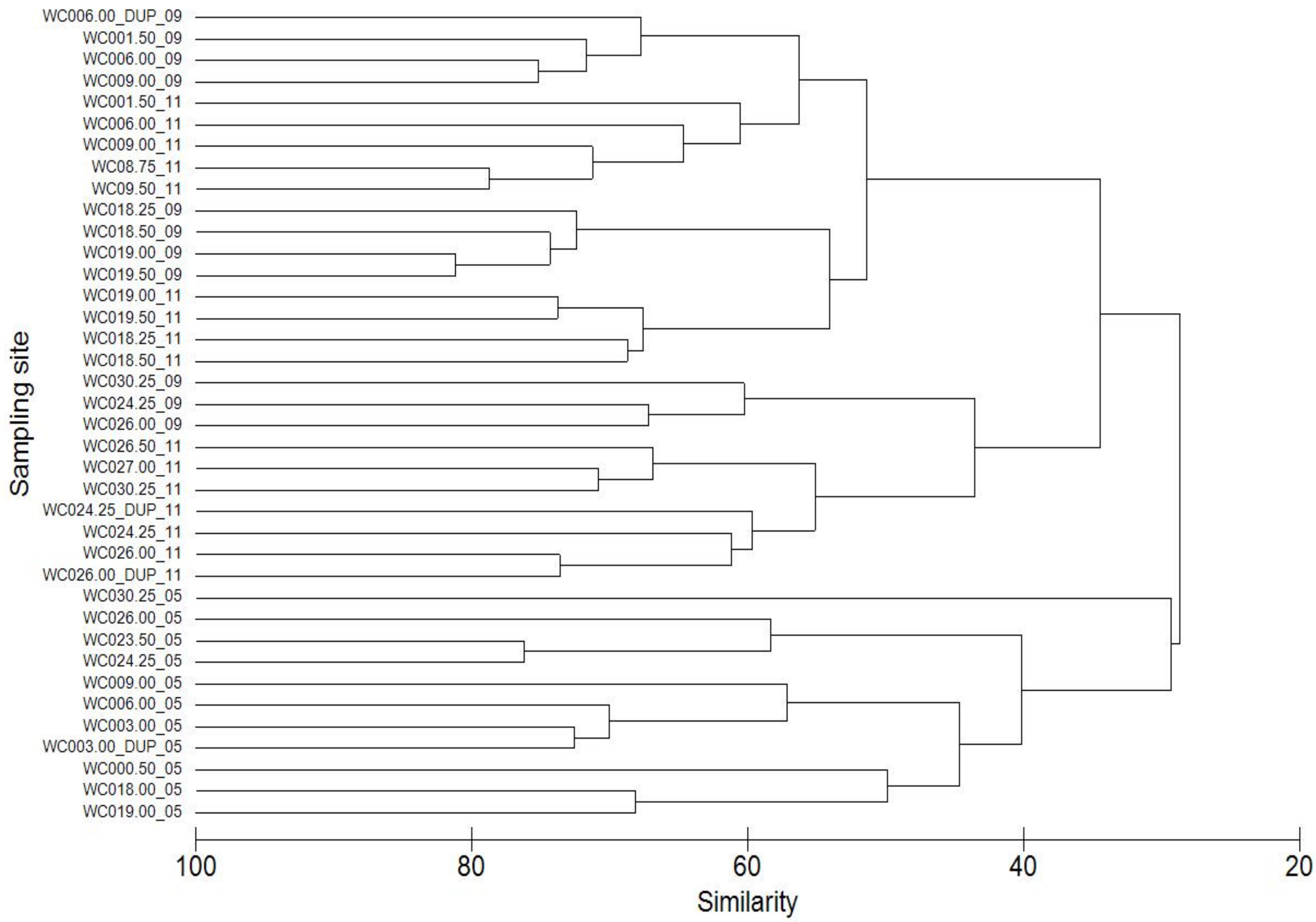


Figure 4. CLUSTER analysis of macroinvertebrate community data from 2005-2011. The final two digits of each site label refer to year. DUP indicates a replicate sample taken for quality control purposes.

Conclusions

Many restoration projects are undertaken with the assumption that creating improved physical habitat automatically results in an increase in biodiversity, which in turn restores impaired or lost ecological processes. This prevalent "the field of dreams hypothesis" (Palmer *et al* 1997) has not been consistently 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 over the long-term, and on a regular repeating basis. 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 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 will it be possible to know if the community composition 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 three times within the past six years, to establish baseline conditions in the stream and assess changes following a variety of habitat restoration projects. Analysis of macroinvertebrate community data collected in 2005, 2009, and 2011 indicates the following:

- The overall composition of the benthic macroinvertebrate community in Whychus Creek changed substantially from 2005 to 2009, but remained more stable from 2009 to 2011.

- Biotic conditions reflected by PREDATOR and IBI scores from 2005 to 2011 show sustained improvement among sites in the downstream reaches. Scores from mid-stream and upstream sites indicate a downward trend in biological condition by 2011, especially among upstream sites.

- Community data from 2011 may be anomalous due to the effects of an unusually snowy winter and cold wet spring, leading to higher faster flows and colder temperatures throughout the season. However, as these data suggest decreased abundance and lower biotic conditions among upstream sampling sites, land use in this area should be examined for any changes from 2009 that might account for diminished stream health.

- Mean % fine sediment optima values for replacement taxa were significantly lower than the mean values for missing taxa from 2005-2011, which may reflect changed conditions following restoration activities. Replacement taxa had significantly lower mean seasonal maximum temperature optima compared to missing taxa only in 2011.

- Although PREDATOR and IBI scores showed a downward trend for upstream sampling sites from 2005 to 2011, these sites are actually improving based on abundance and relative proportions of the sensitive EPT taxa (mayflies, stoneflies, and caddisflies). The mean values for both the proportion of all collected taxa comprised of EPT as well as the relative abundance of EPT individuals were significantly higher in 2011 compared to 2005. This difference may be an artifact of the colder wetter conditions prevailing in winter 2010/spring 2011, although the mean values for these attributes from upstream sites in 2009 suggests a sustained trend towards higher richness and abundance of EPT.

- Monitoring at these sampling sites should be continued at regular intervals in the future. Repeated sampling at the same sites will allow ongoing evaluation of stream biological integrity as the macroinvertebrate community composition changes in response to habitat restoration, especially as instream work required for the planned channel restoration at Camp Polk is likely to have short-term negative effects. Long-term monitoring will also provide an indication of the time span needed for stream macroinvertebrate communities to stabilize, as well as detect any unexpected results of habitat alterations in the biotic community.

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Appendix A. Map of sampling sites along Whychus Creek

Whychus Creek sampling sites, 2005-2011

Dots mark the downstream end of the sampling reach, with the exception of WC00150, WC01825, WC01950, and WC02525, which are marked at the upstream end.

Appendix B. Macroinvertebrate monitoring field datasheet

Site ID ____ Date___________

Sampled by: ____

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

Lat./Long. (decimal degrees): N ___________________________ W __________________________

Location verified by: GPS / Flags / Signs / Roads / Topo map / other (describe):

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/revetment				Landfill/trash		
Buildings				Park/lawn/informal rec.		
Industrial				Row crops		
Rural residential				Pasture/range/hay field		
Urban residential				Livestock w/stream access		
Pavement/cleared lot				Logging within last 5 yrs		
Road/railroad				Mining/sand & gravel		
Pipes (inlet/outlet)				Forest/woodland		
Other:						

Qualitative observations:

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

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

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

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

Type of algae: none / filamentous (strands >2") / close-growing / floating clumps

Physical characteristics:

Substrate

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

Water depth

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

Additional notes or observations (including other wildlife noted):

Appendix C. Sampling site PREDATOR and IBI scores

I. PREDATOR O/E scores.

Observed/expected (O/E) score reflects enriched (>1.23), least disturbed (0.93 – 1.23), moderately disturbed (0.79-0.92), or most disturbed (≤0.78) sites.

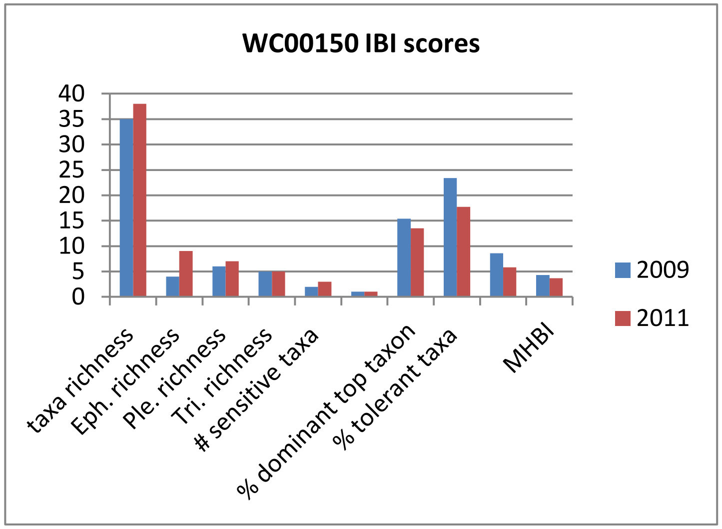
Site	2005	2009	2011	Overall Trend
WC00050	0.832696	N/A	N/A	N/A
WC00150	N/A	0.832498	0.836044	↔
WC00300	0.585774	N/A	N/A	N/A
WC00600	0.668248	0.831966	0.838606	↑
WC00875	N/A	N/A	0.753175	N/A
WC00900	0.648243	0.91514	0.836575	↑
WC00950	N/A	N/A	0.840275	N/A
WC01800	1.079496	N/A	N/A	N/A
WC01825	N/A	0.981688	0.6528	↓
WC01850	N/A	0.89988	0.569871	↓
WC01900	1.239275	0.981938	0.7344	↓
WC01950	N/A	1.0638	0.652978	↓
WC02350	0.815577	N/A	N/A	N/A
WC02425	0.906197	0.820987	0.49403	↓
WC02600	0.815082	0.65668	0.576268	↓
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	↔

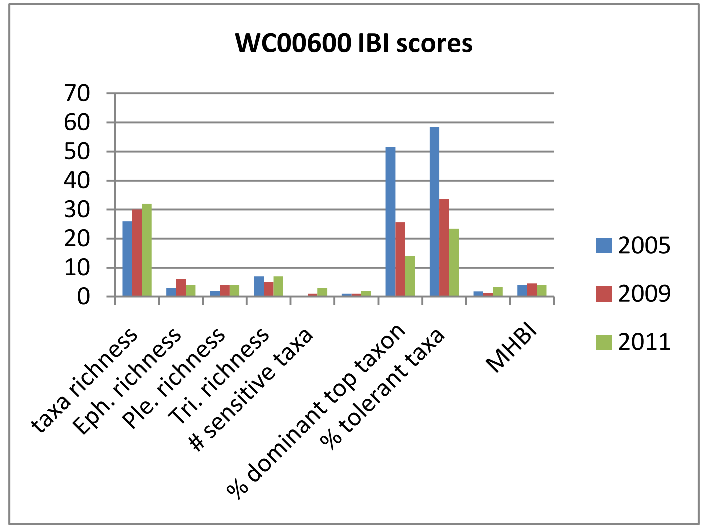
II. IBI scores.

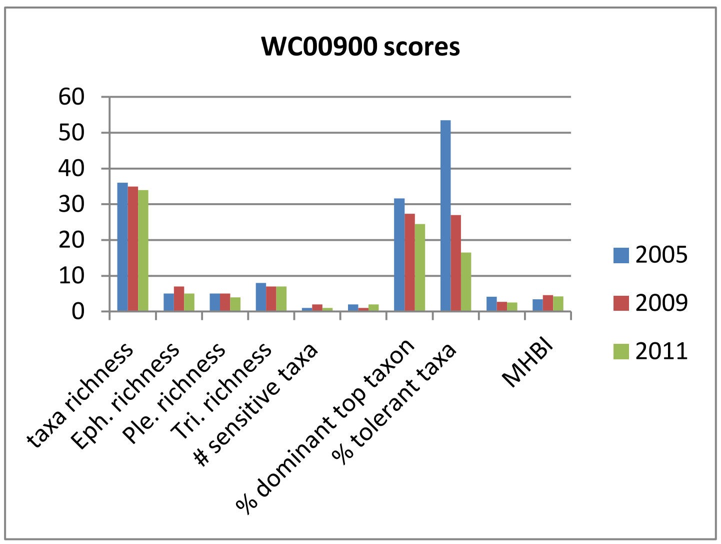
Summed scores reflect minimal (IBI >39), slight (IBI = 30-39), moderate (IBI = 20-29), or severe impairment (IBI <20).

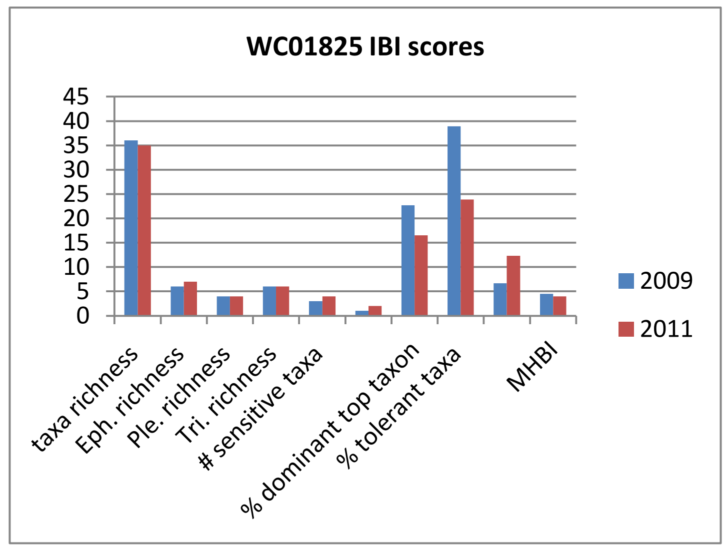
Site	2005	2009	2011	Overall Trend
WC00050	30	N/A	N/A	N/A
WC00150	N/A	36	44	↑
WC00300	26	N/A	N/A	N/A
WC00600	24	30	36	↑
WC00875	N/A	N/A	38	N/A
WC00900	32	32	32	↔
WC00950	N/A	N/A	38	N/A
WC01800	32	N/A	N/A	N/A
WC01825	N/A	34	34	↔
WC01850	N/A	32	22	↓
WC01900	36	30	28	↓
WC01950	N/A	32	34	↔
WC02350	28	N/A	N/A	N/A
WC02425	28	34	26	↔ / ↓
WC02600	28	38	28	↔ / ↓
WC02650	N/A	N/A	32	N/A
WC02700	N/A	N/A	36	N/A
WC03025	34	38	36	↔

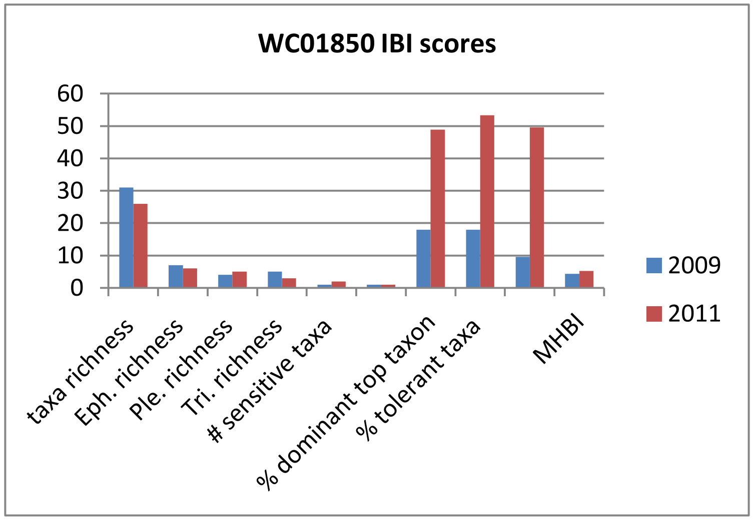
Appendix D. Site-specific changes in IBI metric values across years

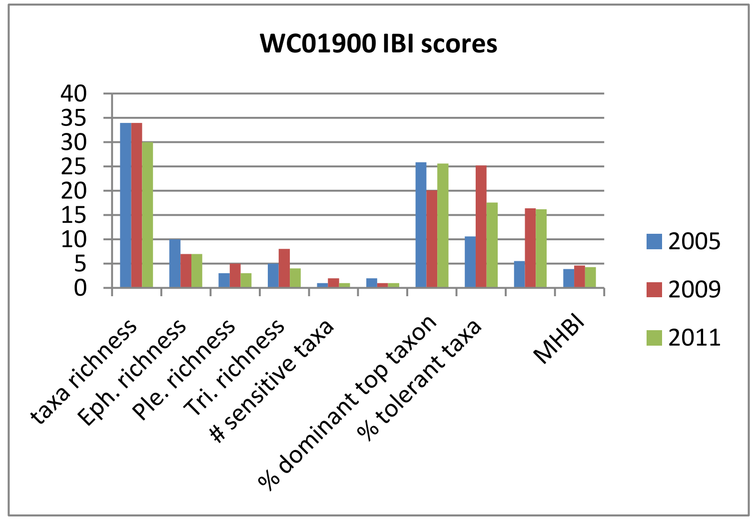


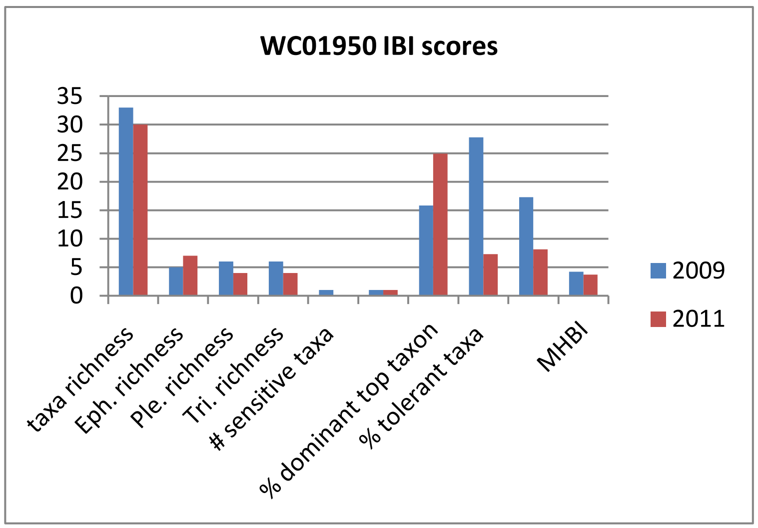


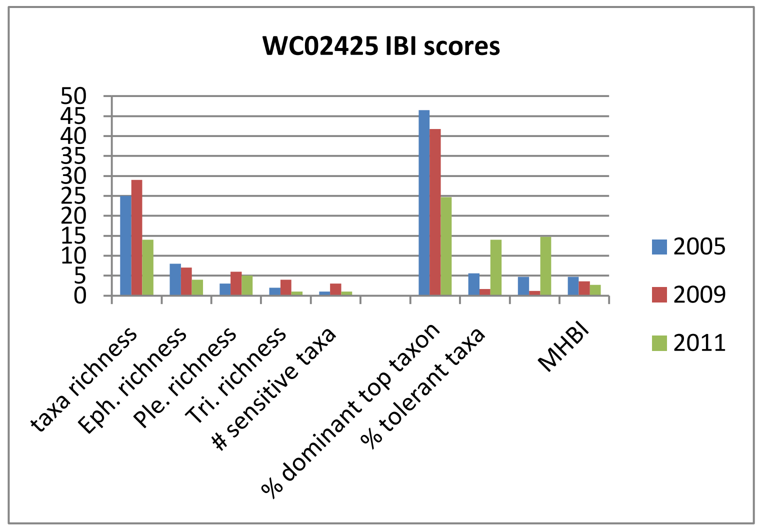


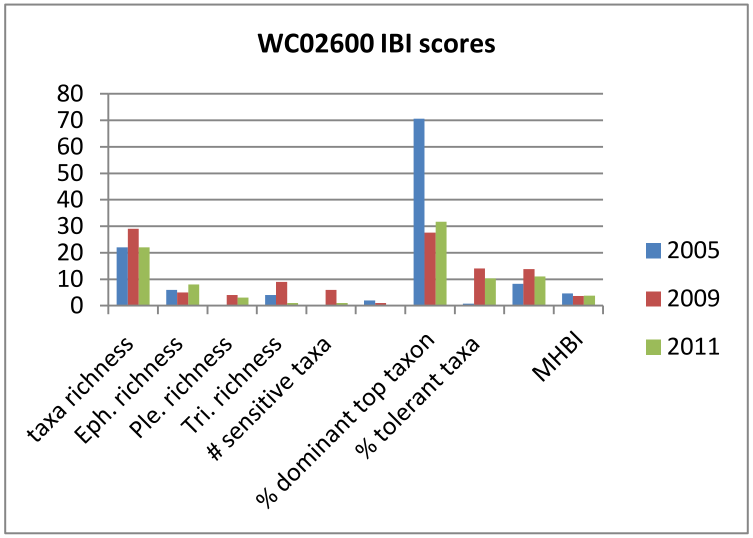


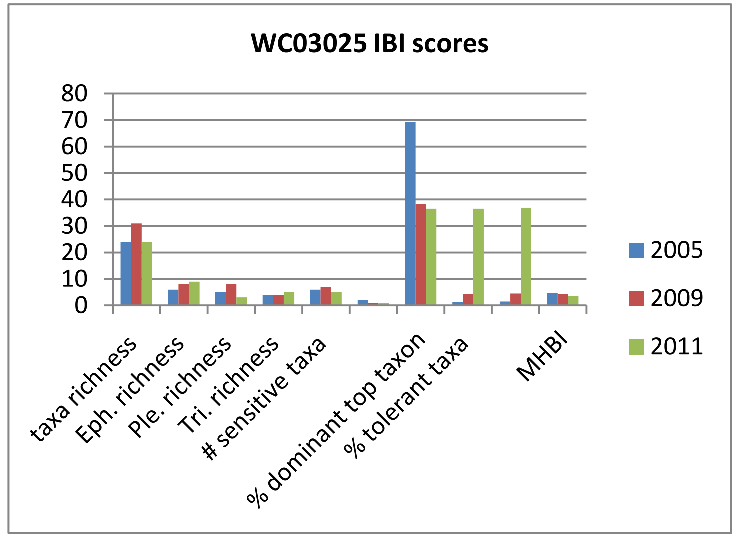

WC01850 IBI scores
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
% dominant top taxon
% tolerant taxa
MHBI
2009
2011


WC01900 IBI scores
40
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
% dominant top taxon
% tolerant taxa
MHBI
2005
2009
2011


WC01950 IBI scores
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
% dominant top taxon
% tolerant taxa
MHBI
2009
2011


WC02425 IBI scores
50
45
40
35
30
25
20
15
10
5
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
% dominant top taxon
% tolerant taxa
MHBI
2005
2009
2011


WC02600 IBI scores
80
70
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
% dominant top taxon
% tolerant taxa
MHBI
2005
2009
2011


WC03025 IBI scores
80
70
60
50
40
30
20
10
0
taxa richness
Eph. richness
Ple. richness
Tri. richness
sensitive taxa
% dominant top taxon
% tolerant taxa
MHBI
2005
2009
2011

Appendix E. Macroinvertebrate Taxa List for Whychus Creek, 2005-2011

Phylum/subphylum	Class/Subclass	Order	Family	Genus	Species	2005	2009	2011
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	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	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	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	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		√		
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 and salmon were reintroduced to Whychus Creek beginning in 2007 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 fish data for Whychus Creek annually to track the status and trends of fish populations. Although as in previous years *O. mykiss* accounted for the majority of fish caught in 2011, survey results suggest a decrease in the population since 2009. *O. mykiss* density ranged from 10-42 fish/100m^2 in five Whychus reaches in 2011, down from 2009 and 2010. Redd counts detected 41 redband redds in 2011, down from 2010 and slightly lower than 2009 numbers. The majority of redds detected have been located in the Alder Springs area in all years sampled. 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

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 also 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. In 2017 and 2022, five and ten years after returning fish are first passed upstream of the hydroelectric project, PGE will conduct genetic analysis to determine relative proportions of juvenile redband and steelhead. 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 2011 native fish monitoring in Whychus Creek, compiled from PGE reports (Hill and Quesada 2012, Quesada et al. 2012). We compare *O. mykiss* population estimates and redd counts from 2011 to 2007 - 2010 results; 2006 native fish monitoring data were collected using different methods and are not comparable to 2007-2010 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 Population Estimates

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 is at Camp Polk at rm 19 (rkm 25.5). 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).

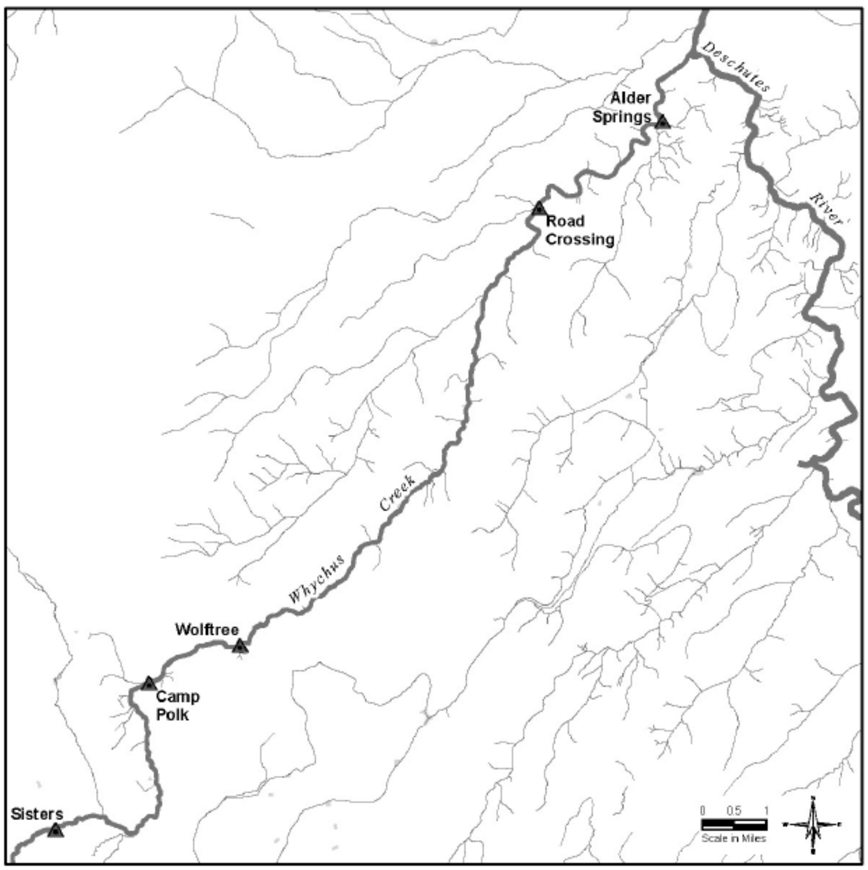

Figure 1.
Study reaches on Whychus Creek for fish population estimates. Alder Springs, Road 6360 Crossing, Camp Polk and Sisters reaches (reaches 1-4) have been surveyed annually since 2006; Wolftree (Reach 5) was sampled in 2009, 2010, and 2011. Reproduced with permission from Quesada *et al* 2012**.**

Fish population sampling was conducted during the low flow period from August 18 to September 24, 2011. Study reach lengths ranged from 73-223 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.

High flows in 2011 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. (2012). All fish captured were recorded by species. *O. mykiss* > 60 mm were 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.

Spring Chinook Juvenile Density

In addition to mark-recapture surveys, beginning in 2009 PGE conducted snorkel surveys at three sites in Whychus Creek to generate density estimates for juvenile Chinook: the 6360 Road Crossing, Wolftree, and at Sisters. The Sisters site was not snorkeled in 2011 because no Chinook fry were released into this reach in 2011. Daytime snorkel surveys were conducted seasonally in spring, summer, and fall. Two snorkelers made two to three upstream passes through each site, with each snorkeler covering an equal portion of the stream during each pass. Snorkelers alternated positions in the stream after each pass to control for bias. Snorkelers recorded fish species, size class, and habitat unit. If after two passes snorkel counts were within 10% of each other, a third pass was not conducted. Density was calculated from snorkel data using a bounded count according to Dambacher (2002).

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, a screwtrap was located on the Upper Deschutes River Arm of Lake Billy Chinook (Figure 2). The trap was operated from February 16th until May 24th, when it broke and was subsequently inoperable. During peak migration (mid-March for Chinook and late May for steelhead) the screwtrap was operated seven days per week and checked daily. During the remainder of the migration season, the trap was checked four days per week. 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.

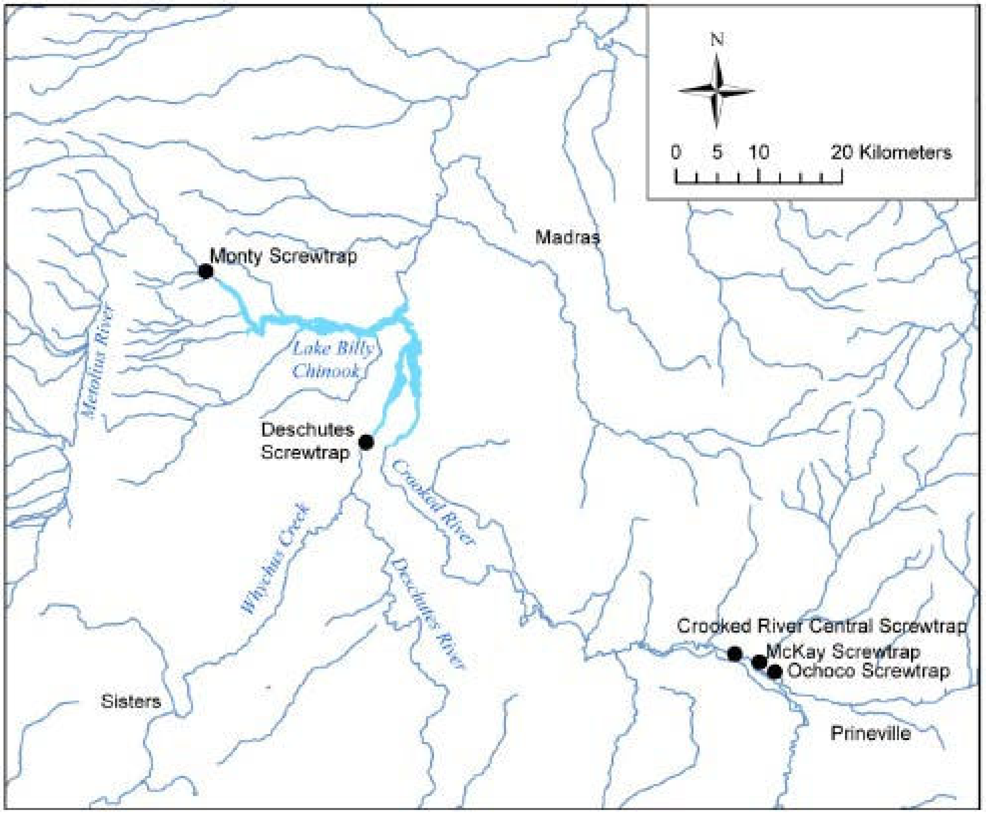

Figure 2.
2011 Screwtrap locations. Reproduced with permission from Hill and Quesada 2012.

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. PGE will conduct 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) will be surveyed every year; each year two additional 1-km will be randomly selected from thirty 1-km reaches between the mouth of Whychus Creek and Sisteers . Four of the ten original reaches will continue to be surveyed annually to help establish a population trend and to identify the temporal and spatial *O. mykiss* spawning distribution.

In 2011 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 upstream of the 6360 Road crossing (rkm 11) and at rkm 34, 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 3). 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.

Beginning in 2009 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 but will increase their efforts to do so in 2012, which will be the last year without steelhead present.

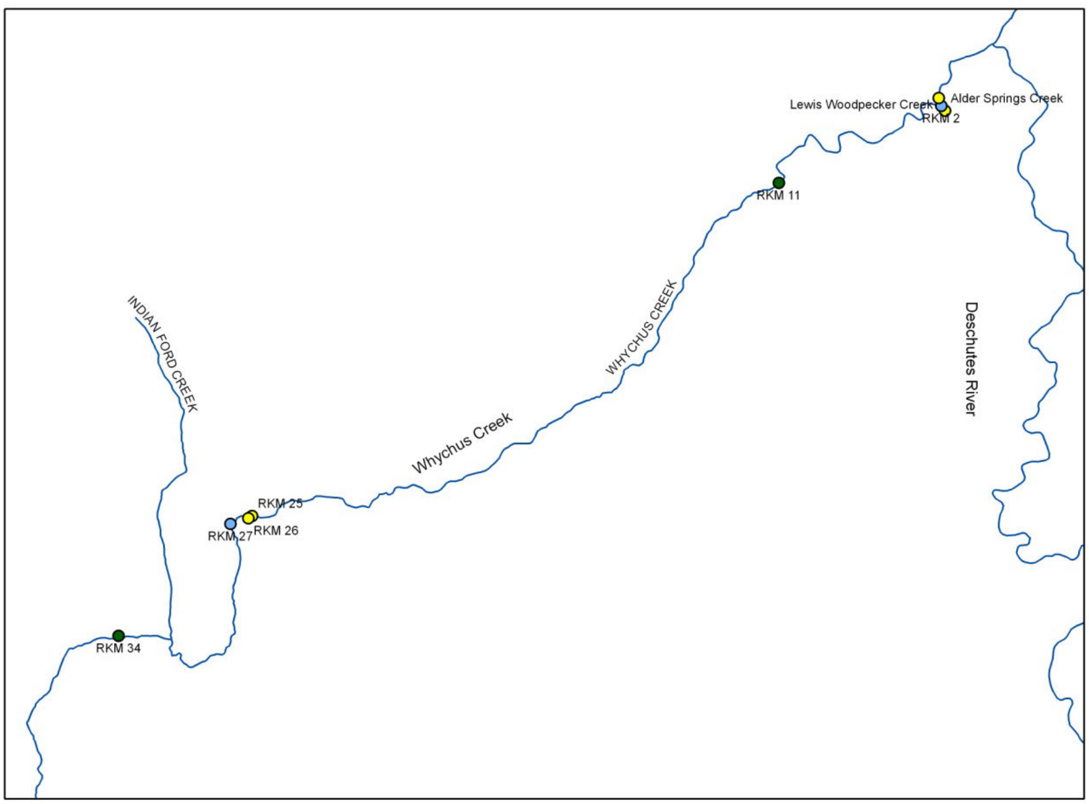

Figure 3.
Redband redds were counted in eight reaches in 2011: two designated index reaches, at rkm 2 (reach 1) and rkm 27 (reach 8), two randomly selected reaches at rkm 11(upstream of the FS Road 6360 crossing) and rkm 34; and in four original reaches (reach 2, Alder Springs Creek; reach 3, Lewis Woodpecker Creek; reach 6, rkm 25; and reach 7, rkm 26) surveyed from 2006-2009. Reproduced with permission from Quesada *et al* 2012.

Results

Species Composition

As in previous years, the majority of fish captured in Whychus Creek in 2011 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 five years of sampling (Table 2). Although steelhead fry have been released every year since 2007 and some proportion of individuals would be expected to remain in the creek during one to two subsequent years of sampling, estimated density for each study reach has fluctuated between years with no consistent pattern between reaches (Figure 4). Density estimates for 2011 are lower than in any previous year surveyed. Densities at Camp Polk in particular appear to have decreased in recent years to levels similar to those observed at other Whychus sites. Size distribution of *O. mykiss* from 2007-2011 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 2007-2011. 2006 data are not comparable, and thus are not included, due to differences in sampling methods.

	O. mykiss/100m²				
Reach	2007	2008	2009	2010	2011
1 (Alder Springs)	48 (± 28)	24 (± 24)	12 (± 4)	11 (± 4)	24 (± 5)
2 (Road 6360)	25 (± 10)	9 (± 3)	24 (± 9)	13 (± 4)	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)
5 (Wolftree)	-	-	21 (±7)	106 (± 29)	42 (± 9)
USFS site at TSID	-	-	2.4 (1.5-4.0)	-	-

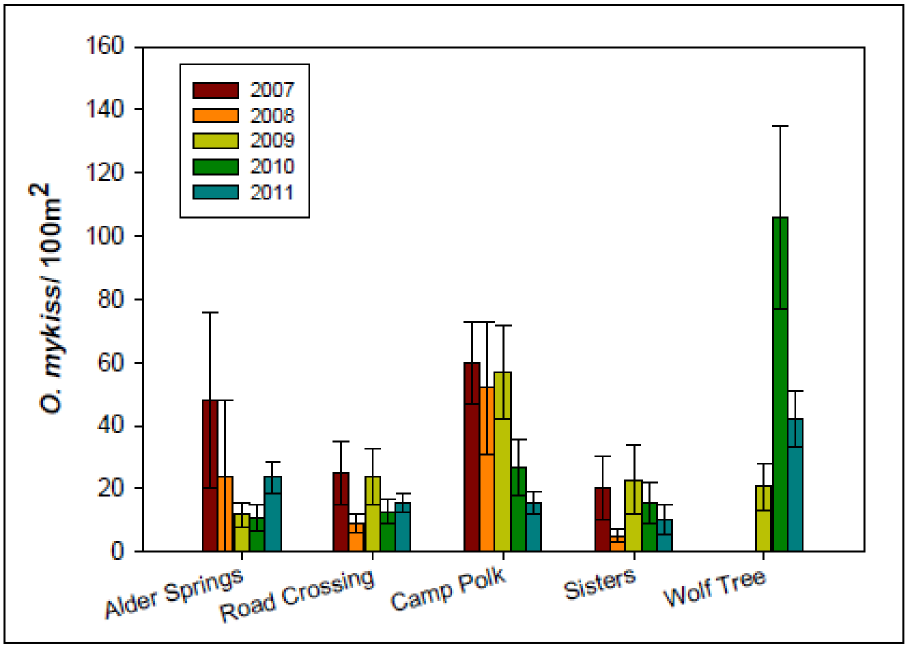

Figure 4.
O. mykiss densities for five Whychus Creek sites from 2007 to 2011. Reproduced with permission from Quesada *et al* 2012.

Spring Chinook Juvenile Density

Mark-recapture surveys resulted in 2011 density estimates of 0-15 juvenile Chinook/100m^2, while 2011 snorkel surveys yielded estimates of only 0-1 Chinook/100m^2 (Table 3). These numbers are down from 2010 and 2009 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	
Reach	Mark Recapture	Snorkel	Mark Recapture	Snorkel	Mark Recapture	Snorkel
1 (Alder Springs)	6 ± 5.8	--	0	--	4 ± 2	--
2 (Road 6360)	4.7 ± 2.3	8	1 ± 0.5	2	2 ± 1	0
3 (Camp Polk)	17.4 ± 14.7	--	0	--	0	--
4 (Sisters)	2 ± 0.8	4	0	4	0	--
5 (Wolftree)	8.3 ± 6.1	0	25 ± 16	4	15 ± 19	1

Smolt production

Despite operating throughout the migration season, PGE was not able to capture enough fish in 2011 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 screwtrap in the thalweg at this location until the reservoir reached full pool (M. Hill personal communication, 2012).

O. mykiss Redd Surveys

As of 2011 no adult steelhead have been passed above the dams and therefore all redds observed during March-July surveys are assumed to be from redband trout; other salmonid species occurring in Whychus Creek are fall-spawning fish. Redband spawning has been documented in Whychus Creek 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. Although surveys were conducted throughout the 2011 spawning season, as in previous years high flows and turbidity limited surveys from late May through July. Surveyors detected a total of 41 redds in 2011, averaging 8 redds/km, down from 65 total redds and 11.5 redds/km in 2010 (Table 4). Also consistent with previous years, the Alder Springs area accounted for over half (51.2%) of all redds observed (Figure 5). Spawning began in April and continued through June, peaking in June.

Table 4. Redds detected by site and year, and totals for each year. “ns” indicates sites not sampled.

	O. mykiss redds/site				
Site	2007	2008	2009	2010	2011
Alder Springs	69	48	24	36	21
Rimrock	38	18	10	ns	ns
Camp Polk	21	8	9	14	11
RKM 8	ns	ns	ns	3	ns
RKM 11	ns	ns	ns	0	3
RKM 23	ns	ns	ns	12	ns
RKM 34	ns	ns	ns	ns	6
Total	**128**	**74**	**43**	**65**	**41**

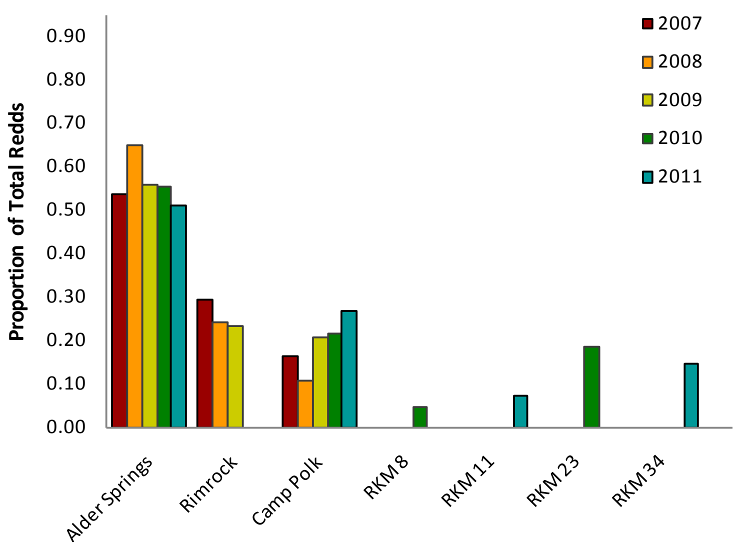

Figure 5.
Proportion of total redband redds detected by year in Whychus Creek. Rimrock sites were not surveyed in 2010 and 2011; the rkm 8 and rkm 23 sites were surveyed only in 2010, and rkm 11 and rkm 34 sites surveyed only in 2011, randomly selected under the rotating panel design.

Discussion

Population Estimates

Despite annual releases averaging 272,600 fry and 5,460 smolts, juvenile *O. mykis*s density remains relatively low in Whychus as compared to densities estimated for the two Crooked River tributary creeks, McKay and Ochoco (Quesada *et al* 2012). Density decreased at Camp Polk and Sisters in every year with the exception of 2008. Density at the 6360 Road Crossing and Alder Springs sites followed the same pattern as Camp Polk and Sisters, but increased in 2011 while density at Camp Polk and Sisters continued to decline. The exception to this trend is the Wolftree site, first surveyed in 2009, where density spiked in 2010 and remained 2-3x higher in 2011 than at any other site. 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 consistently 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 1999, 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 steelhead spawning criteria and low gravel and cobble percentages (Mork 2012), may also be resulting in low juvenile redband abundance. Macroinvertebrate data from 2011 indicated deteriorating conditions in the Camp Polk reach from 2009 to 2011, including a 2011 rating of “most disturbed” under the PREDATOR model (Hubler 2008, Mazzacano 2011). Although the same reach received a “good” habitat quality rating for spawning and emergence and for overwintering life stages and a “fair” rating for summer rearing and migration, habitat quality ratings for this reach reflect 2009 habitat survey data and 2009 habitat conditions, consistent with more favorable 2009 macroinvertebrate ratings. Habitat quality and macroinvertebrate data thus provide some evidence for compromised stream conditions that may be supporting lower numbers of juvenile O. mykiss than 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 and 2011 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. 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 are 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.

Similar to steelhead trends, spring Chinook densities have remained highest and even increased over 2009 numbers at Wolftree, while densities at other sites have fallen dramatically, in keeping with the overall decrease in Chinook density since fry and smolts were first released in 2009. As no Chinook have naturally spawned in Whychus Creek as of 2011, all Chinook observed in Whychus were 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.

High water during sampling continues to reduce the effectiveness of sampling efforts by preventing the use of blocknets, in 2011 resulting in fish moving between sampling reaches 1, 2, and 5 (Alder Springs, 6360 Road Crossing, and Wolftree), and potentially reducing the accuracy of population estimates for these sites. Despite this challenge, PGE researchers have made substantial progress from 2010 to 2011 in refining sampling methods to increase capture efficiency and confidence around density estimates. Precision (95% confidence interval) around estimates of density for Whychus Creek has increased from ±42% in 2007 to ±26% in 2011. PGE attributes improvements in sampling methods to increased reach length, use of 2 electrofishers and continuous direct current (DC), and to having a more experienced crew.

Smolt production estimates

As in previous years, high water impacted screwtrap operations in 2011, preventing PGE from estimating smolt production for the Deschutes. PGE plans to operate screwtraps seven days a week for the duration of the 2012 migration season to increase catch by 30-40% and improve outmigration estimates.

O. mykiss Redd Counts

Redd numbers dropped in 2011 but remain in the range observed in previous years, with the exception of 2007. The high number of redds detected in 2007 corresponds to the only sampling year during which high and turbid flows did not impede surveys. 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. 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 metrics sampled from 2007 to 2011 reveal no clear O. mykiss or spring Chinook population trends, data collected since 2007 generally chart decreases in numbers of fish detected. After falling from 2007 to 2008, average O. mykiss densities increased each year from 2008 to 2010 but fell in 2011 to their lowest point yet. Redband redd numbers have 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. These decreases do not represent a statistically significant trend.
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 and wider wetted width, 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 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 and a larger sample of redd measurements 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, trends, and interactions between the two populations.

Restoration partners originally 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 inadequate 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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