Articles

Trends in the Distribution and Abundance of Yellowstone Cutthroat Trout and Nonnative Trout in Idaho

Kevin A. Meyer,* Erin I. Larson, Christopher L. Sullivan, Brett High

Idaho Department of Fish and Game, 1414 East Locust Lane, Nampa, Idaho 83686

Abstract

The distribution and abundance of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* across their native range is relatively well-known, but evaluations of trends in distribution and abundance over time are lacking. In 2010–2011, we resurveyed 74 stream reaches in the upper Snake River basin of Idaho that were sampled in the 1980s and again in 1999–2000 to evaluate changes in the distribution and abundance of Yellowstone cutthroat trout and nonnative trout over time. Yellowstone cutthroat trout occupied all 74 reaches in the 1980s, 70 reaches in 1999–2000, and 69 reaches in 2010–2011. In comparison, rainbow trout *O. mykiss* and rainbow $\times$ cutthroat hybrid occupancy increased from 23 reaches in the 1980s to 36 reaches in 1999–2000, and then declined back to 23 reaches in 2010–2011. The proportion of reaches occupied by brown trout *Salmo trutta* and brook trout *Salvelinus fontinalis* was largely unchanged across time periods. Yellowstone cutthroat trout abundance declined from a mean of 40.0 fish/100 linear meters of stream in the 1980s to 32.8 fish/100 m in 2010–2011. In contrast, estimates of abundance increased over time for all species of nonnative trout. Population growth rate (λ) was therefore below replacement for Yellowstone cutthroat trout (mean = 0.98) and above replacement for rainbow trout (1.07), brown trout (1.08), and brook trout (1.04), but 90% confidence intervals overlapped unity for all species. However, λ differed statistically from 1.00 within some individual drainages for each species. More pronounced drought conditions in any given year resulted in lower Yellowstone cutthroat trout abundance 1 y later. Our results suggest that over a span of up to 32 y, the distribution and abundance of Yellowstone cutthroat trout in the upper Snake River basin of Idaho appears to be relatively stable, and nonnative trout do not currently appear to be expanding across the basin.

Keywords: cutthroat trout; nonnative trout; population growth rates; status; trends

Received: September 9, 2013; Accepted: May 19, 2014; Published Online Early: June 2014; Published: December 2014

Citation: Meyer KA, Larson EI, Sullivan CL, High B. 2014. Trends in the distribution and abundance of Yellowstone cutthroat trout and nonnative trout in Idaho. *Journal of Fish and Wildlife Management* 5(2): 227–242; e1944-687X. doi: 10.3996/092013-JFWM-062

* Corresponding author: kevin.meyer@idfg.idaho.gov

Introduction

The Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* is one of the most abundant and broadly distributed cutthroat trout subspecies in western North America (Behnke 2002). Nevertheless, their distribution and abundance has declined substantially over the past century because of anthropogenic activities that have considerably altered the ecological river-scape that they occupy (reviewed in Gresswell 2011). Such declines led to a petition in 1998 to list Yellowstone cutthroat trout under the US Endangered Species Act (ESA 1973, as amended), and a court-ordered status review was initiated in 2005, but in both instances the species' status did not warrant ESA 1973 protection (USFWS 2001, 2006). Nevertheless, the Yellowstone cutthroat trout is considered a species of concern by the State of Idaho and other entities, and its status in Idaho is closely monitored by the Idaho Department of Fish and Game (IDFG; e.g., Meyer et al. 2003b, 2006; IDFG 2007).

Nonnative trout pose the largest threat to the long-term persistence of Yellowstone cutthroat trout via two primary vectors: hybridization and competition. First, hatchery rainbow trout *O. mykiss* of coastal origin have been stocked throughout the range of Yellowstone cutthroat trout for >100 y (Gresswell 2011). Because rainbow trout readily hybridize with Yellowstone cutthroat trout, genetic introgression threatens to reduce pure populations of Yellowstone cutthroat trout across much of their range (May et al. 2007). A second threat to Yellowstone cutthroat trout is competition with nonnative brook trout *Salvelinus fontinalis* because they typically outcompete and often displace cutthroat trout populations in western North America (reviewed in Griffith 1988, Dunham et al. 2002, and Fausch et al. 2009). Brook trout have also been stocked across the native range of Yellowstone cutthroat trout, have established numerous self-sustaining populations, and continue to invade new streams (Dunham et al. 2002; Fausch et al. 2009). The brown trout *Salmo trutta* is a third nonnative salmonid species in sympatry with native Yellowstone cutthroat trout in many areas, but the interaction between these two species appears to be relatively benign (Gresswell 2011; but see Quist and Hubert 2005).

Numerous status assessments have been conducted to evaluate the distribution and abundance of Yellowstone cutthroat trout (e.g., Varley and Gresswell 1988; Thurow et al. 1997; Kruse et al. 2000; Meyer et al. 2006; May et al. 2007; Gresswell 2011). However, most status assessments for Yellowstone cutthroat trout that we are aware of have lacked information about trends in abundance. The IDFG conducted broad-scale trend-monitoring that included numerous population electrofishing surveys scattered across several river drainages in the upper Snake River basin of Idaho. Initial surveys were conducted in the 1980s, and the same reaches were sampled again in 1999–2000 (Meyer et al. 2003b). They found that Yellowstone cutthroat trout distribution and abundance at the surveyed reaches were relatively unchanged over the 10–20-y time period, but rainbow trout and hybrids had expanded in distribution. Because more than a decade has passed since the last surveys, our first objective was to repeat the same sampling reaches for a third time to evaluate changes in Yellowstone cutthroat trout occupancy, abundance, and population growth in Idaho. Because of the aforementioned concern posed by nonnative salmonids, we evaluated the same characteristics for rainbow trout, brook trout, and brown trout at these sampling reaches to assess whether these species were expanding in distribution or abundance.

Drought has an almost universally negative effect on stream-dwelling fish populations (Matthews and Marsh-Matthews 2003). Impacts can be 1) immediate, such as with short-term changes in fish populations due to loss of habitat quantity or quality (Magoulick and Kobza 2003) or physiological stress on individuals; or 2) delayed, such as with reduced reproductive success (Elliott et al. 1997). Resident salmonids have been shown to be negatively affected by drought conditions (e.g., Elliott 2000; Hakala and Hartman 2004; White and Rahel 2008), and drought has often been implicated as a primary abiotic factor affecting cutthroat trout populations (Dunham et al. 1999; Haak et al. 2010, Gresswell 2011). Consequently, as a second objective we evaluated whether drought conditions were related to Yellowstone cutthroat trout population growth rates in the upper Snake River basin.

Methods

The upper Snake River basin in eastern Idaho is a high desert region of the Intermountain West with streams that range in elevation from 1,020 m at Shoshone Falls to headwater tributaries near 2,400 m. Shoshone Falls is a 65-m waterfall on the Snake River that forms a natural barrier to upstream invasion by redband trout *O. mykiss gairdnerii*, which are native below the waterfall. Snowmelt drives discharge in most tributaries of the upper Snake River and stream flows normally peak in May and June. However, dams control flows in the larger tributaries for downstream irrigation use, resulting in peak flows being delayed to summer months in these reaches.

More than 100 stream reaches across the upper Snake River basin were originally surveyed by IDFG biologists in the 1980s as index reaches to monitor Yellowstone cutthroat trout populations. Because trout abundance varies greatly with respect to the spatial context of streams (Milner et al. 1993), we made a concerted effort to only resample reaches in later time periods that could be relocated with certainty. This was possible because the original biologists usually created detailed maps of the reaches with descriptive field notes. They also often drove stakes at the bottom and top of the reaches to mark the reach boundaries, and took photos or noted other distinguishing characteristics at the reach. These same IDFG biologists involved in the original sampling from 1980 to 1989 returned in 1999–2000 to help find the surveyor stakes or other features that marked the original reach boundaries. Only those reaches for which survey boundaries could clearly be determined from surveyor's stakes, field notes, maps, and photographs were chosen for resampling. Of the 77 reaches relocated and resurveyed by Meyer et al. (2003b), we were able to resample 74 reaches in 2010–2011 (private property access was denied at 3 of the originally sampled locations).

To control for seasonal variation in trout abundance (Hicks and Watson 1985; Petty et al. 2005), we repeated sampling close to the original calendar date, with more than one-half of repeat surveys occurring within 14 d of the original sampling date (mean = 17 d). Stream reaches that we surveyed in this study ranged from 49 to 7,300 m long, from 2 to 79 m wide, from 1,457 to 2,097 m in elevation, and included first- to seventh-order streams (at a 1:100,000 scale hydrography). Specific conductivity ranged from 136 to 835 μS/cm.

Trout were collected by electrofishing, anesthetized, identified to species, measured for total length to the nearest millimeter, and released. In the smaller, shallower streams (n = 57), we conducted two- or three-pass removal electrofishing using backpack electrofishing units and pulsed direct current at settings of 50–60 Hz, 0.5–2.0 ms pulse width, and 300–800 V. We estimated trout abundance and associated variance using the maximum likelihood model in the MicroFish software package (Van Deventer and Platts 1989). If no trout were captured on the second pass, we considered the catch on the first pass to be the estimated abundance.

At reaches too large to perform removal electrofishing (n = 17), we conducted mark–recapture electrofishing with a canoe- or boat-mounted unit using a pulsed direct current waveform operated at 60 Hz, 400–500 V, and a duty cycle of 20–40%. We marked all trout with a caudal fin clip during the marking run, and we captured marked and unmarked trout during a single recapture run 1–7 d after the marking run. We used the Fisheries Analysis Plus software program to calculate abundance estimates and associated variance using the Lincoln–Petersen mark-recapture model as modified by Chapman (1951). We separated estimates into the smallest size-groups possible (usually 100 mm), which met the criteria that 1) the number of fish marked in the marking run multiplied by the catch in the recapture run was at least four times the estimated population size, and 2) at least three recaptures occurred per size group. This was done to control for size selectivity in the catch data (Reynolds 1996), and because resulting modified Petersen estimates are generally <2% biased (Robson and Regier 1964).

The use of block nets during electrofishing (to ensure the population was closed) was not completely standardized between time periods (Table 2). During all time periods, we never used block nets at reaches where mark–recapture sampling was conducted because the streams were too wide to set block nets. We assumed there was no movement of marked or unmarked fish into or out of the study reach between the mark and recapture runs, and attempted to reduce the likelihood of movement by lengthening the study reaches to 327–7,300 m in length (mean = 3,175 m) and avoiding the release of fish near the reach boundaries during the marking run. About 25% of the depletion reaches were also too wide to set the 5.5 m block nets. For the remaining narrower depletion reaches, block nets were never used in the 1980s, were nearly always used in 1999–2000, and were sporadically used in 2010–2011. We assumed this inconsistency in block net use had minimal influence on our study because previous studies have demonstrated that block nets have little effect on salmonid movement or population estimates in small streams (Young and Schmetterling 2004, 2012).

As a result of low capture efficiencies for small fish in larger rivers, we could not estimate abundance of fish <100 mm in length in the mark–recapture reaches. Also, the length of age-0 fish was inconsistent across reaches and among species. For these reasons, we did not include fish <100 mm in length in any of our estimates of trout abundance. Separating abundance estimates for each species was often not possible because low abundance and limited catch often precluded such partitioning. Therefore, in order to maintain consistency in methodology across reaches and time periods, we pooled all trout species for an overall estimate of trout abundance at the reach scale (e.g., Mullner et al. 1998; Isaak and Hubert 2004; Carrier et al. 2009), and we then calculated point estimates for each species based on the proportion of catch that each species comprised (Meyer and High 2011). Since 2001, hatchery rainbow trout have been sterilized in Idaho to eliminate hybridization concerns (Kozfkay et al. 2006); the few hatchery rainbow trout we encountered during sampling were readily distinguishable from wild rainbow trout based on fin condition and were removed from further consideration in this study.

We differentiated Yellowstone cutthroat trout, rainbow trout, and cutthroat × rainbow hybrids (hereafter, hybrids) using the phenotypic characteristics outlined in Meyer et al. (2003b). In short, we considered any fish with 1) ≤5 spots on top of the head, 2) no white leading edge on the pelvic or anal fins, 3) spots on the body that were large and concentrated posteriorly and dorsally, and 4) a faint or strong throat slash to be Yellowstone cutthroat trout. Rainbow trout and hybrids were clustered into one group for analyses, and we visually identified them by some combination of the presence of white edges on the pelvic or anal fins, smaller spots evenly distributed throughout the body, >5 spots on the top of the head, or absence of a throat slash.

Because stream width was not measured at some reaches for some time periods, we standardized abundance to fish per 100 linear meters of stream. To assess trends in abundance at individual reaches, we used linear regression with sample year as the independent variable and $\log_e$ transformations of trout abundance as the dependent variable. Because the natural logarithm is undefined for zero, we substituted abundance estimates of 0 fish/100 m with 0.0001 fish/100 m. This approach assumes that the population changes in an exponential manner and that the rate of population change is constant over the sampling period; the slope of the regression line is equivalent to the intrinsic rate of change (r) for the population (Gerrodette 1987). We generated point estimates of r at each of the reaches sampled for any species detected in at least one of the sampling periods. We converted each point estimate of r to an estimate of population growth rate (λ) by raising Euler's number (e) to the power of r. We calculated an overall mean λ and an associated variance for each species, and estimated means and variances by species and by drainage. Estimates of λ with 90% confidence intervals (CIs) that overlapped unity (i.e., 1.00) suggest a stable population through time. Those populations with $\lambda < 1.00$ are declining in abundance through time, and those with $\lambda > 1.00$ are increasing.

We assessed whether population growth rates for each species at each reach was associated with several basic stream habitat conditions at that reach. Because most of the reaches (about 75%) had more than one trout species sampled at some point during the study, and reach occupancy fluctuated for all four trout species across time, we focused on stream habitat conditions that have been shown to influence nonnative trout invasions, trout distribution patterns, or competitive interactions between trout species. In particular, elevation (which is often considered a surrogate for stream temperature), wetted width, and stream gradient often influence nonnative salmonid invasion success, mediate competitive interactions among salmonids, and explain partitioning of salmonids along stream networks (e.g., Fausch 1989; Bozek and Hubert 1992; Rahel and Nibbelink 1999; Peterson et al. 2004; Torgersen et al. 2006). Whether these patterns are more often the result of interference competition, or behavioral and (or) physiological responses to habitat requirements or preferences, has not been resolved. Regardless of the causative mechanism, however, we expected that these three stream attributes might influence the population growth rates we observed in the species we encountered.

At each study site, we determined elevation (m) from U.S. Geological Survey 1:24,000-scale topographic maps using Global Positioning System–acquired Universal Transverse Mercator coordinates obtained at the lower end of the reach. We calculated stream wetted width (m) from the average of 10 transects space equally throughout the reach. We determined gradient (%) using the software package All Topo Maps Version 2.1 for Windows (iGage Mapping Corporation, Salt Lake City, UT). The distance (m) between the two contour lines that bounded the study site was traced, and gradient was calculated as the elevational increment between those contours divided by the traced distance.

We treated each reach as a sample unit, and plotted habitat variables against λ for each species to look for data abnormalities and nonlinear relationships (especially wedge-shaped [Terrell et al. 1996] or bell-shaped [Isaak and Hubert 2004] patterns) but none were apparent. Consequently, we evaluated the strength of the habitat–λ relationships using linear regression.

To assess whether drought negatively affected Yellowstone cutthroat trout in our study, we compared their abundance to the Palmer Drought Severity Index (PDSI) computed for southeast Idaho by the National Climatic Data Center (Heddinghaus and Sabol 1991; NOAA National Climatic Data Center, http://www.ncdc.noaa.gov). The PDSI is computed as a monthly value based on a balance between moisture supply, soil characteristics, and evapotranspiration (Palmer 1965). Negative PDSI values of 0 to -0.5 are normal, -0.5 to -1 indicate incipient drought, -1 to -2 indicate mild drought, -2 to -3 indicate moderate drought, -3 to -4 indicate severe drought, and <-4 indicate extreme drought. Positive PDSI values follow a similar qualitative categorization for wet weather. We averaged the 12 monthly values to compute a mean PDSI for the year.

Because drought affects stream flow, which inherently affects stream width, the fish abundance metric we used (fish/100 m) could have potentially been lower (or higher) in some years if stream width was narrower (or wider) and territory size influenced abundance (Grant and Kramer 1990). To account for this, we transformed abundance data to fish/100 m^2 (without the $\log_e$ transformation), and consequently discarded data at 9 of the 74 reaches where width measurements were not available for all three surveys through time.

Abundance estimates for each of the three sampling periods at a reach were then normalized to a z-score based on the mean abundance at the reach across all sampling periods, so that each reach had a mean abundance z-score of zero and a standard deviation of one. This normalizing of the abundance data had the effect of making all reaches contribute equally to the abundance–drought relationship rather than hinging more heavily on the reaches with the highest abundance. For each year of fish sampling, we estimated a mean z-score for all reaches surveyed in that year, and related mean annual PDSI to mean annual z-scores for the same year using linear regression. We surveyed fish abundance in 12 separate years; therefore, this gave us a sample size of 12 for this analysis. Because drought could have potentially affected recruitment or had other delayed impacts that outweighed effects on within-year abundance, we related drought to Yellowstone cutthroat trout abundance at time lags from 1–4 y because most cutthroat trout in eastern Idaho are $\leq$4 y old (Meyer et al. 2003a). We examined residuals of the linear regression model for outliers, influential data points, unequal variance, and lack of normality, but none of these issues were apparent in the model results.

We used SAS (SAS Institute Inc. 2009) to perform all statistical analyses. Throughout our analyses, we used a significance level of $\alpha = 0.10$ to increase the power of detecting trends or differences between time periods (Peterman 1990; Maxell 1999; Dauwalter et al. 2009).

Results

Yellowstone cutthroat trout occupied all 74 reaches in the 1980s, compared with 70 reaches in 1999–2000, and 69 reaches in 2010–2011 (Table 1). In comparison, rainbow trout and hybrid occupancy increased from 23 reaches in the 1980s to 36 reaches in 1999–2000, and then declined back to 23 reaches in 2010–2011. The decline in rainbow trout and hybrid occupancy from 1999–2000 to 2010–2011 occurred in four of the seven drainages. The number of reaches occupied by brown trout and brook trout was generally unchanged across all time periods, but for both species, several reaches occupied in the 1980s were unoccupied in 2010–2011 and several reaches unoccupied in the 1980s were occupied in 2010–2011 (Table 2).

Yellowstone cutthroat trout abundance increased from an average of 40.0 fish/100 m in the 1980s to 42.0 fish/100 m in 1999–2000, then declined to 32.8 fish/100 m in 2010–2011 (Table 2; Figure 1). Considering reaches individually, abundance was lower from the 1980s to 2010–2011 at 39 of the 74 reaches and higher at 35 reaches. Yellowstone cutthroat trout comprised 80% of the abundance of all trout in the 1980s, 76% in 1999–2000, and 69% in 2010–2011.

Abundance of rainbow trout and hybrids remained relatively unchanged for all time periods, with mean abundance (in reaches where they were present during at least one survey) of 8.9, 7.1, and 9.8 fish/100 m in the

Table 1. Number of reaches occupied (by fish ≥ 100 mm TL) by species and drainage during electrofishing surveys in the 1980s, 1999–2000 and 2010–2011 in the upper Snake River basin of Idaho.

		Drainages							
Metric	**Period**	**Raft River/ Goose Creek**	**Portneuf River**	**Blackfoot River**	**Willow Creek**	**South Fork Snake River**	**Palisades Reservoir/ Salt River**	**Teton River**	**Total**
Number of sites surveyed		4	10	9	5	16	26	4	74
Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*									
Number of sites occupied in:	1980s	4	10	9	5	16	26	4	74
	1999–2000	2	9	9	5	16	25	4	70
	2010–2011	2	9	9	3	16	26	4	69
Rainbow trout *Oncorhynchus mykiss* and hybrids									
Number of sites occupied in:	1980s	1	8	2	2	4	2	4	23
	1999–2000	1	8	8	0	11	4	4	36
	2010–2011	2	4	3	0	8	2	4	23
Brown trout *Salmo trutta*									
Number of sites occupied in:	1980s	0	0	0	3	5	11	1	20
	1999–2000	0	1	0	2	7	8	0	18
	2010–2011	0	2	0	1	8	9	2	22
Brook trout *Salvelinus fontinalis*									
Number of sites occupied in:	1980s	1	1	7	0	0	1	4	14
	1999–2000	2	0	2	0	0	0	4	8
	2010–2011	2	0	5	1	0	0	4	12

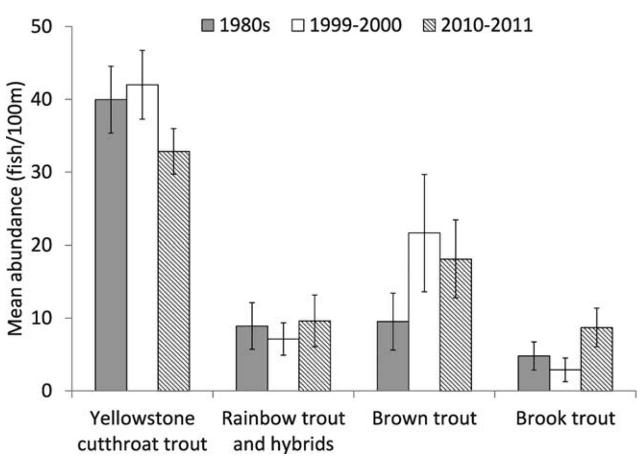

Figure 1. Estimates of mean abundance (±1 SE) by species during electrofishing surveys in the 1980s, 1999–2000, and 2010–2011 at 74 stream reaches in the upper Snake River basin of Idaho. Estimates only included reaches where a particular species was present in at least one time period.

1980s, 1999–2000, and 2010–2011, respectively (Table 2; Figure 1). Brown trout abundance increased from the 1980s (mean = 9.5 fish/100 m) to 1999–2000 (21.6 fish/100 m) and then declined slightly by 2010–2011 (18.1 fish/100 m). Brook trout abundance remained relatively unchanged from the 1980s (4.8 fish/100 m) to 1999–2000 (2.9 fish/100 m) but rose from 1999–2000 to 2010–2011 (8.7 fish/100 m).

Although abundance of Yellowstone cutthroat trout showed a numerical decline from 1999–2000 to 2010–2011, this did not translate to a negative population growth rate across the entire time period in most instances. Across all 74 reaches, mean λ = 0.98 for Yellowstone cutthroat trout and 90% CIs overlapped unity (0.96–1.00; Table 3). Within individual drainages, Yellowstone cutthroat trout population growth rate was declining in the Willow Creek (mean λ = 0.85; 90% CI = 0.73–0.97) and Teton River (mean λ = 0.98; 90% CI = 0.97–0.99) drainages, and in no drainages was population growth rate increasing.

Table 2. Trout abundance (fish/100 m) by species and drainage during electrofishing surveys in the 1980s, 1999–2000 and 2010–2011 in the upper Snake River basin of Idaho.

Reach	Stream	Zone 12 UTMs at downstream reach boundary		Reach length (m)	Reach gradient (%)	Mean width (m)	Estimate method[1]	Year of estimate		
		Easting	Northing					1980s	1999–2000	2010–2011
				Raft River/Goose Creek drainages						
1	Birch Creek	261034	4652363	96	3.0	1.5	D	1987	2000[2]	2010
2	Cold Creek	262182	4665819	72	5.5	2.1	D	1987	2000[2]	2010[2]
3	Eightmile Creek	321035	4668961	83	4.1	1.1	D	1986	2000[2]	2010
4	Trout Creek	733972	4658472	120	1.0	2.5	D	1987	2000[2]	2010
				Portneuf River drainage						
5	Pebble Creek	417656	4732605	208	0.9	2.3	D	1986	2000	2011
6	Pebble Creek	413858	4731734	104	3.9	4.7	D	1986	1999[2]	2011
7	Pebble Creek	412900	4732600	106	1.9	3.6	D	1986	1999[2]	2011
8	Pebble Creek NF	412890	4733081	133	1.9	1.8	D	1986	1999[2]	2011
9	Big Springs Creek	410333	4735083	105	4.5	4.4	D	1986	1999[2]	2011
10	King Creek	414280	4739567	76	2.1	8.1	D	1986	2000	2011
11	Toponce Creek	416977	4744482	88	1.1	6.0	D	1986	2000	2011
12	Toponce Creek, MF	413335	4747494	73	2.3	4.7	D	1986	2000[2]	2010
13	Toponce Creek, SF	412865	4745074	101	2.3	3.4	D	1987	2000[2]	2011[2]
14	Toponce Creek, SF	412687	4744985	113	1.7	6.0	D	1987	2000[2]	2011[2]
				Blackfoot River drainage						
15	Blackfoot River	454738	4740964	4,347	0.2	12.1	D/MR	1988	2000	2011
16	Blackfoot River	471051	4738792	1,698	0.2	16.3	MR	1988	2000	2011
17	Blackfoot River	472916	4740978	1,753	<0.1	11.4	D/MR	1988	2000	2011
18	Diamond Creek	479079	4735907	183	0.6	5.1	D	1988	2000	2011
19	Diamond Creek	481636	4732056	153	0.9	5.6	D	1980	2000[2]	2011[2]
20	Diamond Creek	482174	4730544	167	0.6	3.7	D	1987	2000[2]	2011
21	Diamond Creek	482524	4729938	87	0.6	3.5	D	1987	2000[2]	2011
22	Diamond Creek	483346	4727783	66	0.4	2.5	D	1987	2000[2]	2011[2]
23	Diamond Creek	483605	4727421	162	1.3	3.4	D	1988	2000[2]	2011[2]
				Willow Creek drainage						
24	Willow Creek	438006	4801125	859	0.3	8.8	MR	1984	2000	2011
25	Willow Creek	437008	4795659	561	4.2	10.8	MR/D	1984	2000	2011
26	Brockman Creek	455047	4784591	93	0.2	5.7	D	1983	2000	2010
27	Corral Creek	463166	4785905	76	1.3	1.3	D	1982	2000[2]	2011
28	Corral Creek	463257	4786240	134	1.4	1.9	D	1982	2000[2]	2011
				South Fork Snake River drainage						
29	Snake River	413677	4845870	7,300	<0.1	79.0	MR	1988	2000	2011
30	Snake River, SF	428214	4844051	4,800	0.1	46.0	MR	1989	1999	2011
31	Snake River, SF	437379	4835945	2,900	0.2	66.0	MR	1989	2000	2011
32	Snake River, SF	465305	4814032	4,900	0.2	71.0	MR	1989	1999	2011
33	Burns Creek	461714	4827654	82	2.3	5.9	D	1980	2000	2011
34	Burns Creek	486690	4806929	86	2.0	5.3	D	1980	2000	2011
35	Pine Creek	471082	4817372	66	1.6	11.1	D	1980	2000	2011
36	Pine Creek	475034	4820470	90	0.7	9.3	D	1988	2000	2011
37	Pine Creek	475120	4820535	76	0.7	6.0	D	1980	2000	2011
38	Pine Creek	476201	4822256	80	0.9	4.8	D	1988	2000	2011
39	Pine Creek, NF	477731	4823094	72	1.4	5.3	D	1982	2000	2010
40	Pine Creek, NF	478169	4825839	80	1.5	7.9	D	1981	2000	2011
41	Rainey Creek	478491	4811570	159	1.5	5.7	D	1980	2000	2010

Table 2. Extended.

Trout abundance (fish/100 m)											
Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*			**Rainbow trout *Oncorhynchus mykiss* and hybrids**			**Brown trout *Salmo trutta***			**Brook trout *Salvelinus fontinalis***		
1980s	**1999–2000**	**2010–2011**	**1980s**	**1999–2000**	**2010–2011**	**1980s**	**1999–2000**	**2010–2011**	**1980s**	**1999–2000**	**2010–2011**
1.3	0.0	0.0							1.3	30.6	27.5
5.0	0.0	0.0	0.0	0.0	100.6				0.0	6.3	5.0
6.1	23.2	29.3									
3.9	6.9	18.7	50.8	0.9	3.1						
43.9	14.0	2.6	14.0	1.0	4.7				1.0	0.0	0.0
32.5	44.8	44.9	6.3	7.1	0.0						
75.8	44.2	34.9	4.0	13.5	7.4						
34.7	15.1	6.0									
23.9	25.8	3.8	4.8	2.0	0.0						
7.1	0.0	0.0									
7.5	4.5	10.4	104.9	1.1	1.2	0.0	29.2	27.8			
2.5	17.4	33.8	76.3	78.3	12.5	0.0	0.0	2.5			
115.5	46.0	19.2	18.8	1.0	0.0						
29.2	146.9	52.2	1.8	7.1	0.0						
0.8	12.5	32.0	0.0	1.9	0.7				0.0	0.0	0.1
15.1	36.6	32.7	0.2	7.0	0.0				0.0	0.1	0.7
5.9	13.2	101.5	0.1	6.3	0.0				0.1	0.5	14.0
8.4	10.6	19.1							1.1	0.0	9.3
130.4	61.9	10.9	0.0	6.8	0.0				4.8	0.0	6.1
24.7	64.2	30.3	0.0	6.7	0.0				2.7	0.0	0.0
9.2	43.7	26.4	0.0	4.6	3.4				2.3	0.0	0.0
50.7	43.9	10.7	0.0	12.2	0.0				1.3	0.0	0.0
17.6	29.1	10.3	0.0	19.0	1.8				5.4	0.0	0.0
21.3	3.9	30.9	1.2	0.0	0.0	2.6	0.6	0.0			
66.9	22.9	40.3	0.7	0.0	0.0	24.2	2.1	0.2	0.0	0.0	0.2
7.7	26.9	0.0				2.0	0.0	0.0			
64.8	6.3	0.0									
27.6	14.6	3.1									
7.5	8.3	6.2	2.9	3.1	3.1	10.0	29.7	40.6			
14.9	34.3	33.4	0.2	0.7	3.9	28.2	113.5	96.1			
46.2	69.8	27.1	0.3	2.0	5.1	45.2	175.7	93.1			
161.0	184.7	122.5	6.3	65.4	119.0	19.1	51.2	79.6			
56.5	33.4	11.8	0.0	20.2	5.3	0.0	0.0	9.2			
7.0	31.4	10.5	0.0	2.3	0.0	0.0	3.5	2.3			
51.5	71.2	76.6	0.0	6.1	21.9						
77.8	24.4	58.9	0.0	0.0	26.7						
155.4	146.8	64.9	0.0	5.1	0.0						
53.8	82.5	56.3	0.0	7.5	0.0						
22.3	18.1	30.6									
43.8	8.8	5.0	0.0	1.2	0.0						
1.3	37.5	14.3				0.0	1.9	10.0			

Table 2. Continued.

Reach	Stream	Zone 12 UTMs at downstream reach boundary		Reach length (m)	Reach gradient (%)	Mean width (m)	Estimate method[1]	Year of estimate		
		Easting	Northing					1980s	1999–2000	2010–2011
42	Rainey Creek	478513	4811828	124	1.7	6.5	D	1980	2000	2011
43	Rainey Creek	481836	4813772	174	0.5	7.6	D	1980	2000	2010
44	Fall Creek	464733	4805719	130	0.8	6.2	D	1988	2000	2011
	Palisades Reservoir/Salt River drainages									
45	Bear Creek	481607	4791560	211	0.7	8.8	D	1980	2000	2011
46	Elk Creek	481792	4790811	146	2.8	4.1	D	1980	2000[2]	2011
47	Big Elk Creek	491079	4796890	106	5.6	6.9	D	1980	2000	2010
48	Big Elk Creek	493081	4797837	148	1.7	7.9	D	1980	2000	2010
49	McCoy Creek	487648	4780309	373	0.7	9.2	MR	1986	2000	2010
50	McCoy Creek	483801	4778223	396	1.1	8.8	MR	1986	2000	2010
51	McCoy Creek	476710	4770800	148	1.0	3.2	D	1986	1999[2]	2010
52	Jensen Creek	487569	4780625	81	2.3	3.3	D	1986	1999[2]	2010
53	Fish Creek	485578	4777936	86	3.8	3.6	D	1986	1999[2]	2010
54	Fish Creek	485809	4775782	92	2.6	3.1	D	1986	1999	2010
55	Barnes Creek	472268	4776309	100	4.9	3.0	D	1986	1999[2]	2010
56	Barnes Creek	472335	4775758	77	4.9	3.2	D	1986	1999[2]	2010[2]
57	Clear Creek	476730	4778890	124	0.9	3.4	D	1986	1999[2]	2010[2]
58	Iowa Creek	479814	4777204	101	2.3	4.0	D	1986	1999[2]	2010[2]
59	Jackknife Creek	491880	4766150	109	0.6	6.1	D	1987	1999	2010
60	Tincup Creek	491945	4761078	153	1.2	5.9	D	1987	1999[2]	2010
61	Tincup Creek	486398	4758462	123	1.3	6.8	D	1987	1999	2010
62	Tincup Creek	491945	4761078	101	1.9	5.1	D	1987	1999[2]	2010
63	Bear Canyon Creek	484010	4758430	61	5.1	1.6	D	1987	1999[2]	2010
64	Stump Creek	493844	4737765	454	0.3	7.0	MR	1986	2000	2011
65	Horse Creek	493760	4737880	86	2.3	2.2	D	1986	1999[2]	2011
66	Crow Creek	489676	4715833	327	0.4	5.4	MR	1986	2000	2010
67	Crow Creek	486157	4709556	112	1.2	3.6	D	1986	1999[2]	2010
68	Sage Creek	491700	4718400	206	0.5	5.3	D	1987	1999[2]	2010
69	Deer Creek	488970	4714550	158	1.3	4.2	D	1986	1999[2]	2010[2]
70	White Dugway Creek	486192	4709482	84	1.3	1.6	D	1986	1999[2]	2010
	Teton River drainage									
71	Teton River	481805	4850358	4,900	<0.1	26.0	MR	1987	1999	2011
72	Teton River	483128	4847608	5,500	<0.1	34.4	MR	1987	2000	2011
73	Teton River	483537	4844388	7,100	<0.1	34.8	MR	1987	2000	2011
74	Teton River	484839	4841139	5,800	<0.1	42.4	MR	1987	1999	2011
	Average									

[1] D is depletion method, MR is mark-recapture method.
[2] Sites where block nets were used.

In comparison, estimates of mean λ across all reaches were above 1.00 for all nonnative salmonids, although 90% CIs overlapped unity in all cases (Table 3). However, increasing or decreasing population growth rates were evident in some drainages. For rainbow trout and hybrids, λ was declining in the Portneuf River drainage (mean λ = 0.79; 90% CIs = 0.69–0.89) and the Willow Creek drainage (mean λ = 0.80; 90% CIs = 0.65–0.95), and increasing in the South Fork Snake River drainage (mean λ = 1.23; 90% CIs = 1.12–1.34). For brown trout, λ was declining in the Willow Creek drainage (mean λ = 0.74; 90% CIs = 0.65–0.83) and increasing in the Portneuf River drainage (mean λ = 1.58; 90% CIs = 1.42–1.74) and the South Fork Snake River drainage (mean λ = 1.19; 95% CIs = 1.08–1.30). For brook trout, λ did not differ from unity in any drainage.

Population growth rates at individual stream reaches were rarely correlated with the stream habitat conditions that we measured at that reach (Table 4). In fact, the only statistically significant associations were a positive

Table 2. Continued Extended.

Trout abundance (fish/100 m)											
Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*			Rainbow trout *Oncorhynchus mykiss* and hybrids			Brown trout *Salmo trutta*			Brook trout *Salvelinus fontinalis*		
1980s	1999–2000	2010–2011	1980s	1999–2000	2010–2011	1980s	1999–2000	2010–2011	1980s	1999–2000	2010–2011
6.5	4.1	22.8									
9.0	40.0	38.3	0.0	1.7	9.0	4.2	11.0	18.6			
12.8	60.5	71.4									
17.9	71.6	56.7	0.0	0.4	0.0	0.0	0.0	3.1			
24.7	36.3	47.3									
8.2	33.0	17.5									
19.9	61.3	31.5									
72.3	35.5	59.2	0.0	0.3	0.0	1.1	0.8	0.0			
107.7	117.5	102.0				1.0	0.0	0.3			
52.9	56.9	72.2									
162.6	0.0	59.2									
48.4	167.0	39.2									
43.5	90.2	34.8									
24.2	33.6	28.3									
7.9	10.5	23.7	0.0	0.0	1.3						
61.5	31.1	41.8									
25.8	31.3	41.2									
29.0	14.0	43.8				0.9	0.0	1.0			
62.7	76.0	31.0	0.7	0.0	0.0	0.7	1.4	0.0			
129.0	64.1	39.3	0.9	0.0	0.9	2.7	0.0	0.0			
66.0	21.0	18.0									
87.8	32.3	1.9	0.0	1.9	0.0						
44.2	123.6	32.5				9.5	25.5	16.9	0.7	0.0	0.0
40.6	70.7	33.7				0.0	4.6	9.3			
3.6	10.4	16.0				5.2	32.1	17.3			
83.6	117.1	71.5				0.9	9.0	15.9			
19.4	31.1	33.2				100.1	108.5	46.8			
37.1	78.6	109.6				7.6	5.7	15.3			
13.1	6.0	36.9				1.2	0.0	0.0	0.0	0.0	7.1
11.0	8.5	6.2	50.2	7.1	40.2	0.0	0.0	0.6	10.6	3.5	14.8
15.7	12.4	13.5	31.2	6.9	33.3	0.1	0.0	0.3	13.4	3.0	17.0
24.3	20.2	11.0	18.8	9.3	16.0				11.2	4.7	23.1
45.1	10.9	23.2	5.8	2.2	12.4				35.1	6.4	40.8
40.0	42.0	32.8	8.9	7.1	9.8	9.5	21.6	18.1	4.8	2.9	8.7

correlation for brook trout ($r = 0.45$) and a negative correlation for Yellowstone cutthroat trout ($r = -0.23$) with respect to reach gradient.

Mean PDSI over the entire time period (1980–2011) was 0.36, and during the years of fish sampling, PDSI averaged 0.59 (Figure 2). The PDSI oscillated from a wet period from 1980 to 1986, to a dry period from 1987 to 1990, to a wet period from 1995 to 1999, and back to a dry period from 2000 to 2004.

The mean annual z-scores of Yellowstone cutthroat trout abundance were most strongly correlated to mean annual PDSI at a 1-y time lag and formed a statistically significant linear relationship ($F = 6.56$; $df = 11$; $r^2 = 0.40$; $P = 0.02$; Figure 3). The relationship indicated that more pronounced drought conditions in any given year resulted in lower Yellowstone cutthroat trout abundance 1 y later. In this relationship, all but one of the data points demonstrated above-average cutthroat trout abundance when PDSI was >0 (wetter than normal) and below-average abundance when PDSI was <0 (drier than normal). All other time lags (including no lag) produced much

Table 3. Mean, 90% confidence intervals (CIs), and range of population growth rates (λ) by species and drainage based on electrofishing surveys in the 1980s, 1999–2000 and 2010–2011 in the upper Snake River basin of Idaho. Estimates in bold indicate statistical significance (at $\alpha = 0.10$).

		Mean population growth rates (λ)							
		Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*		Rainbow trout *Oncorhynchus mykiss* and hybrids		Brown trout *Salmo trutta*		Brook trout *Salvelinus fontinalis*	
Drainage	Number of reaches surveyed	Est. ± 90% CIs	Range	Est. ± 90% CIs	Range	Est. ± 90% CIs	Range	Est. ± 90% CIs	Range
Raft River and Goose Creek	4	0.85 ± 0.21	0.61–1.07	1.32 ± 0.73	0.61–1.07	None captured		1.39 ± 0.40	1.15–1.63
Portneuf River	10	0.95 ± 0.07	0.63–1.12	**0.79 ± 0.10**	0.63–1.12	**1.58 ± 0.16**	1.48–1.68	None captured	
Blackfoot River	9	1.03 ± 0.04	0.93–1.18	1.15 ± 0.20	0.93–1.18	None captured		0.96 ± 0.18	0.62–1.47
Willow Creek	5	**0.85 ± 0.12**	0.65–1.01	**0.80 ± 0.15**	0.65–1.01	**0.74 ± 0.09**	0.68–0.84	None captured	
South Fork Snake River	16	1.01 ± 0.02	0.93–1.10	**1.23 ± 0.11**	0.93–1.10	**1.19 ± 0.11**	1.03–1.49	None captured	
Palisades Reservoir and Salt River	26	1.00 ± 0.01	0.85–1.07	1.04 ± 0.14	0.85–1.07	0.98 ± 0.13	0.64–1.59	1.13 ± 0.72	0.69–1.57
Teton River	4	**0.98 ± 0.01**	0.97–0.99	1.00 ± 0.02	0.97–0.99	1.23 ± 0.34	1.02–1.43	1.01 ± 0.01	1.01–1.03
Total	74	0.98 ± 0.02	0.61–1.18	1.07 ± 0.07	0.61–1.18	1.08 ± 0.09	0.64–1.68	1.04 ± 0.12	0.62–1.63

weaker and nonsignificant relationships with PDSI ($r^2 < 0.12$; $P > 0.27$).

Discussion

Results from this study suggest that Yellowstone cutthroat trout continue to dominate the reaches in our study that were originally established in the 1980s to monitor cutthroat trout populations in the upper Snake River basin. Indeed, occupancy was generally unaltered (down only 7%) and population growth rates across all reaches did not differ from replacement. Although mean abundance across all reaches declined by 18%, this may simply reflect normal temporal fluctuations in trout populations (Dauwalter et al. 2009). Finally, the extent of Yellowstone cutthroat trout allopatry (relative to nonnative salmonids) was similar in 2010–2011 (42% of reaches) as in the 1980s (45%).

Despite these positive findings, there are also causes for concern. For instance, Yellowstone cutthroat trout are no longer present in 5 of the 74 reaches, including in Corral Creek (in the Willow Creek drainage) where they were formerly quite abundant. Second, Yellowstone cutthroat trout abundance in the 1980s was <10 fish/100 m at four of the five reaches no longer occupied by cutthroat trout, suggesting that future extirpations will perhaps be more likely at the eight reaches occupied by cutthroat trout in 2010–2011 where abundance was <10 fish/100 m. Of the five reaches where cutthroat trout were extirpated, nonnative trout increased markedly at two reaches and no trout were captured of any species at three reaches. These findings concur with the general consensus that nonnative trout and habitat alteration are two of the biggest threats to continued range contraction for Yellowstone cutthroat trout across their range (Gresswell 2011). Fortunately, nonnative trout do not appear to be expanding dramatically in the upper Snake River basin, at least in the reaches we surveyed. Although they experienced a 50% increase in total abundance from the 1980s to 2010–2011 across all reaches, nonnative trout currently constitute only about 30% of the abundance of all trout at these reaches.

Table 4. Correlations between trout population growth rates (λ) and reach width, elevation, and gradient at 74 stream reaches surveyed in the 1980s, 1999–2000 and 2010–2011 in the upper Snake River basin of Idaho. Estimates in bold indicate statistical significance (at $\alpha = 0.10$).

Species	Reach width (m)	Reach elevation (m)	Reach gradient (%)
Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*	0.08	0.03	**−0.23**
Rainbow trout *Oncorhynchus mykiss* and hybrids	−0.01	−0.04	0.02
Brook trout *Salvelinus fontinalis*	0.07	−0.25	**0.45**
Brown trout *Salmo trutta*	−0.02	−0.19	0.22
All non-native trout	0.01	−0.19	0.16
All trout	0.13	−0.12	−0.01

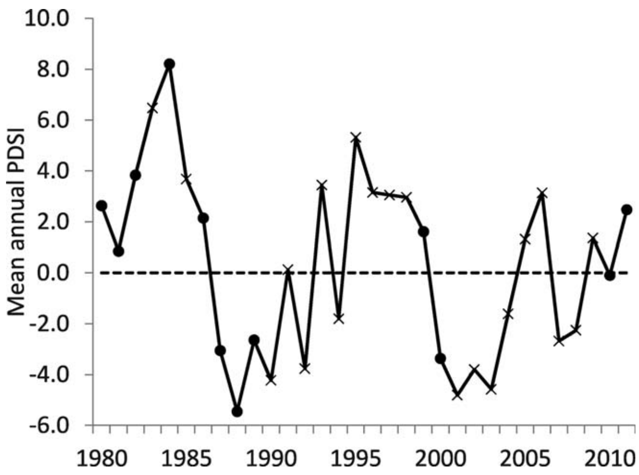

Figure 2. Mean annual Palmer Drought Severity Index (PDSI) computed for southeast Idaho during the study period of 1980 to 2011, during which we resurveyed stream reaches in the upper Snake River basin of Idaho to evaluate changes in the distribution and abundance of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* and nonnative trout over time. Filled circles indicate years when fish sampling occurred.

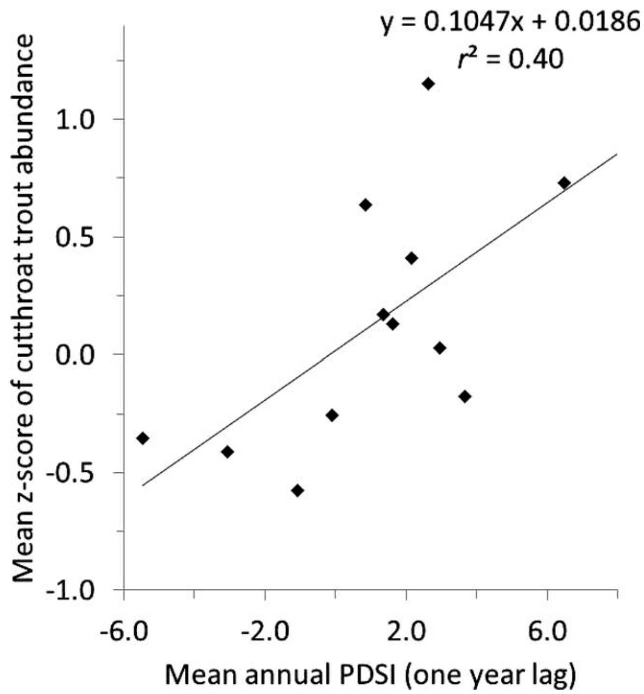

Figure 3. Relationship between mean annual Palmer Drought Severity Index (PDSI) in a given year and mean annual z-scores of the abundance of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* 1 y later at reaches surveyed in the 1980s, 1999–2000, and 2010–2011 in the upper Snake River basin of Idaho.

Rainbow trout and hybrid occupancy expanded from the 1980s to 1999–2000, but then contracted to 1980s levels by 2010–2011. Most of this early expansion and later contraction occurred in the 12 reaches sampled in Diamond Creek (Blackfoot River drainage) and Pine Creek (South Fork Snake River drainage), with rainbow trout and hybrid occupancy in these two streams increasing from zero reaches in the 1980s to 9 in 1999–2000, and then declining to four reaches in 2010–2011. However, even excluding these two streams, occupancy by rainbow trout and hybrids contracted from 27 reaches in 1999–2000 to 19 reaches in 2010–2011. These contractions were not likely caused by misidentifying fish because hatchery rainbow trout are readily distinguishable from wild fish based on fin condition, and wild rainbow trout and hybrids are readily distinguishable from Yellowstone cutthroat trout based on phenotype (Campbell et al. 2002; Meyer et al. 2006). In fact, in a recent study, IDFG biologists correctly identified fish phenotypically (as confirmed by genotyping the fish) as either 1) Yellowstone cutthroat trout or 2) rainbow trout and hybrids for 322 of 353 fish (91%); nearly all of the errors were in mistakenly identifying cutthroat trout as hybrids, or vice versa (K. A. Meyer, unpublished data). Considering this degree of accuracy for individual fish, the likelihood that we entirely missed the presence of rainbow trout or hybrids at the reach scale is quite small.

The decline in occupancy by rainbow trout and hybrids in our study reaches since the 1999–2000 surveys coincides with a decision in 2001 by IDFG to discontinue stocking of catchable-sized hatchery rainbow trout in streams that support Yellowstone cutthroat trout populations, and where stocking presently continues in other streams, to only stock rainbow trout that have been sterilized by pressure treatment to eliminate hybridization issues (Kozfkay et al. 2006; IDFG 2007). Nevertheless, once hybridization is occurring in a population, the proportion of introgression in the population generally increases (Gresswell 2011), likely because spawn timing overlaps very little between rainbow trout and cutthroat trout, but hybrids have a much greater overlap with both species (Muhlfeld et al. 2009). In fact, we are not aware of any studies that have shown a declining level of cutthroat $\times$ rainbow introgression in populations over time. Continuing the above-mentioned hatchery stocking policy and expanding management actions to control or remove rainbow trout and hybrids from streams in the upper Snake River basin (e.g., IDFG 2007; High 2010) may help limit the expansion of rainbow trout introgression in Idaho streams occupied by native Yellowstone cutthroat trout.

Also of concern for long-term conservation of Yellowstone cutthroat trout in the upper Snake River basin is sympatric interactions with brook trout because they so frequently displace native cutthroat trout populations in western North America (e.g., Peterson and Fausch 2003; Peterson et al. 2004; Shepard 2004). However, our results revealed surprisingly little evidence that brook trout were negatively affecting Yellowstone cutthroat trout in the reaches we surveyed, at least in regard to cutthroat trout occupancy and abundance. For example, brook trout were sympatric with Yellowstone cutthroat trout in 14 reaches in the 1980s, and in only one reach were cutthroat trout absent in 2010–2011, whereas brook trout were absent from six of these reaches in 2010–2011. This surprising finding suggests that even established brook trout populations in the 1980s did not always persist. An alternative explanation could be that some brook trout populations in the 1980s were being bolstered by hatchery stocking, but a query of the IDFG stocking database revealed that no brook trout were being stocked at that time in any streams included in our study. The only noteworthy associations between fish abundance and stream habitat conditions were that stream gradient was negatively related to Yellowstone cutthroat trout population growth but positively related to brook trout population growth, suggesting that brook trout may have outcompeted cutthroat trout in the high-gradient reaches. However, if brook trout displace cutthroat trout, it is typically in lower elevation, lower gradient reaches (Hilderbrand 1998; Peterson et al. 2004). Despite the general lack of impact that brook trout appear to have had on Yellowstone cutthroat trout in the reaches we sampled, the vast majority of studies have found that over the long term, brook trout will negatively affect cutthroat trout (Griffith 1988; Dunham et al. 2002; Fausch et al. 2009). The IDFG therefore continues to remove brook trout from Idaho streams where feasible to restore native Yellowstone cutthroat trout in streams where they have been extirpated (IDFG 2007).

Our results suggest that drought severity in any given year has a negative effect on Yellowstone cutthroat trout abundance the following year. Considering that age-0 fish in 1 y were large enough the following year to be included in our abundance estimates, and they would likely have constituted the most abundant age class in most instances, the negative relationship between drought and Yellowstone cutthroat trout abundance is perhaps the result of poor survival or production of age-0 fish during low-flow years. Such an effect on age-0 survival could arise from a number of mechanisms, including reduced reproductive success (Elliott et al. 1997); reduced habitat quality and availability (Hakala and Hartman 2004); poorer food resources for newly emerged fry (Cowx et al. 1984); intensified predation as age-0 fish are forced into closer proximity to predators because of less available space (Larimore et al. 1959); or lower winter flows, which may reduce overwinter survival (Hakala and Hartman 2004). Regardless of the mechanism, the negative effect of drought on Yellowstone cutthroat trout abundance observed here suggests that if drought conditions become more severe in the future (Luce and Holden 2009), declines in Yellowstone cutthroat trout abundance may result.

All three trout species present at the reaches surveyed in the Willow Creek drainage showed statistically significant declines in population growth (Table 3). In the 1990s, the Natural Resource Conservation Service identified the Willow Creek drainage as one of the ten worst soil erosion areas in the United States, and nearly every stream in the drainage is listed as an impaired water on the Clean Water Act's 303d list for excessive sedimentation (Thompson 2004). Although 49% of the watershed has been part of a conservation program during the past 30 y (Thompson 2004), stream habitat conditions remain degraded. Fortunately, there is virtually no cutthroat $\times$ rainbow hybridization in the Willow Creek drainage (Meyer et al. 2006), which has probably enabled Yellowstone cutthroat trout to persist under pervasively poor habitat conditions. However, the long-term viability of Yellowstone cutthroat trout is tenuous in such degraded habitat, especially if stream temperatures in the drainage warm in the future (Isaak et al. 2011).

Our study had a number of important limitations. The primary shortcoming was the nonrandom nature of reach selection by biologists in the 1980s. Reaches were typically established near roads, bridges, culverts, or other accessible features; therefore, they may not be representative of conditions across the landscape (Kadmon et al. 2004). However, our sample size was quite large, the surveyed reaches were broadly distributed across the study area, and they encompassed a wide variety of physical habitat conditions; hence, they are likely minimally biased in regard to spatial coverage, despite the nonrandom nature of site selection (Thompson and Lee 2000; Kadmon et al. 2003; Wagner et al. 2007). Despite the obvious importance of random sampling to ensure that observations are drawn from the population of interest (Garton et al. 2012), trend-monitoring studies for salmonids are usually conducted at index reaches established similarly to those in our study, and rarely is random sampling of the population rigorously adhered to (e.g., Freeman et al. 1988; House 1995; Gowan and Fausch 1996; Waters 1999; Ham and Pearsons 2000). Courbios et al. (2008) note that despite inherent limitations, continuing to monitor index reaches that were not randomly established remains valuable because of the temporal extent of data at such reaches and the ability to examine long-term population dynamics. Moreover, it could be argued that by concentrating study reaches in accessible areas—where detrimental stream alterations are typically more prevalent, and where streams have long been accessible to fish-stocking trucks—our results may represent a worst-case scenario relative to the status of Yellowstone cutthroat trout. In fact, because all 74 reaches contained Yellowstone cutthroat trout in the 1980s, our study design could only have detected range contraction (not expansion). We recommend combining these reaches with randomly sampled reaches such as in Meyer et al. (2006) to allow continued use of this long-term data set as part of a more rigorous study design (Courbios et al. 2008).

With only three surveys over time at each reach, it is difficult to draw decisive temporal conclusions from our results. In fact, the coefficient of variation in Yellowstone cutthroat trout abundance in our study (63%) typifies temporal fluctuations frequently exhibited by trout populations across North America (Dauwalter et al. 2009). As such, the fluctuations we observed may not represent true changes in abundance, but instead may simply reflect interannual variation in population size that typifies many trout populations. Such fluctuations are a common problem when monitoring salmonid population abundance through time because they limit the ability to detect real population trends at a statistically significant level (Dauwalter et al. 2009; Pearsons and Temple 2010). This highlights the need to perhaps monitor these reaches more frequently in the future to ascertain whether population growth rates truly differ from replacement, or whether they are simply oscillating around a relatively stable mean.

If a meaningful number of fish were consistently leaving the reaches without block nets in the 1980s and in 2010–2011, then trout abundance in these two time periods would have been underestimated for the reaches sampled in the smallest streams (about one-half of the reaches). However, we do not believe that the block net differences affected our results appreciably for several reasons. First, the original site boundaries for small streams were always placed at natural breaks in habitat units such as where riffles began, so that swift stream-flow would presumably discourage fish movement out of the reach (Edwards et al. 2003). Second, as Bohlin et al. (1998) also noted, we saw no concentration of salmonids in the vicinity of the block nets in the reaches where nets were used, suggesting that fish were not being pushed long distances by the electrofishing crews. Third, there was no pattern of increasing or decreasing abundance for any species related to whether or not block nets had been used. Finally, a recent study in small streams—comparable in size to our reaches where block net use was inconsistent—used radiotagged trout and shocked in an upstream direction to measure how far trout move in response to backpack electrofishing (Young and Schmetterling 2012). They found that 50% of the tagged trout moved $<$4.5 m and 95% moved $<$18 m. If it is assumed that trout were distributed throughout our depletion reaches (which averaged 113 m), it is unlikely that $>$1–2% of the trout in our surveyed reaches would have moved out of the reach when block nets were not used.

In conclusion, the results of this study suggest that Yellowstone cutthroat trout populations in the upper Snake River basin of Idaho may be declining slightly, although at a rate so slow as to not differ statistically from zero change in growth. Our sample reaches were not chosen at random, so caution should be used in extrapolating these results outside of the reaches that were sampled. However, our reaches encompassed a broad geographic range and a wide variety of habitats, so they likely provide a reasonable index of Yellowstone cutthroat trout and nonnative trout population trends in eastern Idaho. Further trend-monitoring over time may help distinguish normal population fluctuations from actual changes in the distribution and abundance of trout populations in the upper Snake River basin. Coupling these reaches with repeat sampling of already established randomly located reaches (such as from Meyer et al. 2006) may help validate any changes detected by repeating our study in the future.

Supplemental Material

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Reference S1. Haak AL, Williams JE, Isaak D, Todd A, Muhlfeld C, Kershner JL, Gresswell R, Hostetler S, Neville HM. 2010. The potential influence of changing climate on the persistence of salmonids of the inland west. U.S. Geological Survey Open-File Report 2010-1236.

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Acknowledgments

Thanks are due especially to Tim Burton, Chip Corsi, Steve Elle, Scott Grunder, Eric Herrera, Tony Lamansky, Virgil Moore, Dan Schill, Bill Schrader, and Russ Thurow for data collection and (or) site relocation over the course of this study. Dan Schill, Jeff Dillon, Dan Garren, Dan Dauwalter, and two anonymous reviewers provided helpful suggestions that improved earlier versions of the paper. The study was funded by the Sportfish Restoration Act.

Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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