Assessment of Macroinvertebrates in the Henry's Fork, 2015-2025

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March 23, 2026

Summary

To address angler concerns about decreased hatches, particularly at Last Chance and the upper Ranch, and holistically assess ecological function throughout the mainstem Henry's Fork, we implemented a statistically rigorous, quantitative study of aquatic invertebrates in 2015. This followed implementation of a watershed-wide water-quality monitoring program in 2014 and prompted subsequent expansion of stream gaging. The result is a set of 56 independent observations of macroinvertebrates collected at six different sites on the river over an 11-year period, each accompanied by a full suite of water-quality and streamflow variables. The sites—Flat Rock, Last Chance, Osborne Bridge, Marysville, Ashton Dam, and St. Anthony—represent conditions over the 80 miles of river most popular among anglers. We analyzed invertebrate abundance, five standard community metrics—Shannon's diversity, EPT taxa richness (the number of mayfly, stonefly, and caddisfly species), Hilsenhoff Biotic Index (HBI), percent non-insects, and percent EPT—as well as the abundance of Pale Morning Duns (*Ephemerella* sp.), *Drunella* mayflies (Flavs + Green Drakes), and Spotted Sedge caddis (Hydropsychidae). For each of these response variables, we tested for dependence on distance downstream from Big Springs as predicted by the River Continuum Concept, difference across sites independent of the river continuum, and dependence on seven streamflow and water-quality variables: annual streamflow, 3-day maximum streamflow, 21-day minimum streamflow, annual flow variability, suspended sediment concentration, conductivity, 7-day maximum water temperature, and total phosphorus. Primary results are:

1. Invertebrate communities in the Henry's Fork are abundant, diverse, and dominated by mayflies, stoneflies and caddisflies; are as good or better than on other western trout streams; and are as good or better than they were on the Henry's Fork decades ago. HBI indicates good to excellent water quality from headwaters to St. Anthony, with little evidence of pollutants.
2. Abundance and %non-insects showed no variability across location along the river and no dependence on any of the streamflow or water quality variables.
3. The other seven responses were best explained by location on the river, with little to no dependence on streamflow or water quality, after accounting for location.
4. Total invertebrate abundance and Pale Morning Dun abundance have decreased significantly since 2015, with marginal evidence for decreases in %non-insects and community diversity.
5. However, HBI, %EPT, and EPT taxa richness have all improved significantly since 2015, indicating overall improvement in water quality and aquatic habitat and possible replacement of non-insect and other degradation-tolerant species with more sensitive and desirable EPT species.
6. There is no evidence that trout populations are limited by invertebrate numbers.
7. It is possible that large changes in PMD abundance and community metrics occurred at Last Chance prior to this study, as a result of increasing temperatures relative to previous decades.
8. However, decreased dry-fly fishing quality at Last Chance/upper Ranch over the past decade is likely due to a combination of lower trout populations, increased turbidity, and altered mayfly emergence behavior due to warmer temperatures, not to decreased insect numbers.

Report format

This report is written in a hybrid style that is aimed at both anglers and scientists. Background information and interpretation of results are tailored toward anglers, with examples relevant to fishing experience. The writing style is conversational, and I have not included literature citations, as would be done in a scientific paper. However, I have included full details on sampling, data compilation, and statistical methods, as well as tables with results of the statistical analyses, for full transparency and for scrutiny by scientists who may read this. For ease of reading the text, I have included all figures and tables at the end of the document.

Why analyze aquatic invertebrates?

Whenever we ask anglers what they value most about their fishing experience on the Henry's Fork, the top answer is invariably the opportunity to fish to rising fish. While this is especially true on the Harriman State Park ("Ranch") reach of the river, it is true on other reaches as well. Whether it is a Green Drake hatch at Flat Rock, Pale Morning Duns at Last Chance, Mother's Day caddis at Ora Bridge, or Gray Drakes on the lower river, the prolific hatches of mayflies, stoneflies, and caddisflies up and down the river are what put the Henry's Fork on the global trout fishing map.

Two ingredients are needed to produce rising fish: fish and emerging insects. Decades of research and monitoring by Idaho Department of Fish and Game, HFF and other partners are unequivocal about the "fish" ingredient. By river reach, the trout population upstream of Island Park Reservoir is determined by water quality in the reservoir and by reservoir volume, the population between Island Park Dam and Riverside is determined by winter outflow from Island Park Dam, and the population downstream of St. Anthony is determined by summertime streamflow. There is no doubt that water supply and management determine trout populations in these reaches, which is why the centerpiece of HFF's work is to provide water users and managers the science and data necessary to keep as much water in Island Park Reservoir as possible all year, maximize outflow from Island Park Dam during the winter, and maintain stable streamflow downstream of St. Anthony at a scientifically determined summertime target that maintains ecological function while keeping as much water in Island Park Reservoir as possible. Everywhere else (essentially Riverside to St. Anthony), trout populations are stable and vary little from year to year, regardless of water supply. And, regardless of river reach, we have never seen any evidence that trout growth is limited by food supply. Despite the large amount of angler concern we receive about streamflow, water quality, river crowding, trout numbers, and hatches, we consistently hear from anglers—even when fishing is tough—that the fish they do catch are always fat and healthy. If food were limiting fisheries anywhere in the river, we would see large numbers of skinny fish. Instead, we consistently see trout in good condition, with numbers driven by water supply.

So, while we are very confident in our knowledge of what drives trout numbers in the Henry's Fork, we are much less certain about what drives insect hatches. To help get some answers, we implemented a scientifically rigorous long-term project in 2015 to monitor aquatic insects and other invertebrates in the Henry's Fork. In 2025, we collected our 11th year of samples.

How do we collect aquatic invertebrates?

Sampling locations

At the start of the program, we recruited a regional aquatic invertebrate expert, Brett Marshall, to oversee field data collection and to conduct our laboratory analysis. Brett has over 40 years of experience in aquatic ecology, specifically in the sampling and analysis of aquatic invertebrates as indicators of stream health. Brett has owned and managed a commercial laboratory in Bozeman since 2007, providing services to government agencies and nongovernmental organizations throughout the western U.S. With Brett's input, we selected five permanent sites on the Henry's Fork that were of interest to anglers and represented the range of habitat conditions found on the river between its headwaters at Big Springs and the confluence of the North Fork Teton River. These five locations are Flat Rock, Last Chance, Osborne Bridge, Marysville, and St. Anthony. We added a sixth site downstream of Ashton Dam to monitor any effects of the Ora Bridge construction that took place in 2020 and periodically sample that site. We have collected samples every year for 11 years at Flat Rock, Last Chance and Osborne Bridge, every year but 2020 (due to covid) at Marysville and St. Anthony, and in 2019, 2021 and 2025 at Ashton Dam. This has given us a sample of 56 observations that represent conditions over the past decade along 80 miles of the river, not coincidentally the same 80 miles referred to in the subtitle of Mike Lawson's book "Fly Fishing Guide to the Henry's Fork".

At the same time as we began designing the invertebrate sampling procedure, we implemented a water-quality monitoring program centered around continuous-recording water quality sondes. More recently, we expanded our stream gaging and hydrologic data compilation programs so that we now have water quality and streamflow data to accompany the invertebrate data at all six locations. This allows us to see whether aquatic insects and other invertebrates respond to changes in streamflow and water quality. See Table 1 and the accompanying map (Figure 1) for more details on the sampling locations. Sonde locations and data can be viewed on our water quality website at https://henrysforkdata.shinyapps.io/scientific_website/, and daily streamflow data can be viewed on our water-quantity site at https://henrysforkdata.shinyapps.io/WaterQuantity/.

Sample timing

Our sample timing is based on the fact that the vast majority of aquatic insects in the Henry's Fork have a one-year life cycle. Adults emerge at a given time during the year, they mate and lay eggs shortly after emergence, the eggs hatch shortly after that, and the nymphs spend the next 350 days or so in the river before emerging as adults at the same time the following year. So, we collect samples on March 16 every year (plus or minus a day or two), which is several months after the last mayfly hatches of the fall and immediately prior to the first mayfly hatches of the spring. This means that the nymphs of all species that live in the river are large enough to be collected in the samples. If we selected any other time to do the sampling, we would miss species that had just recently laid eggs. For example, if we sampled in early July, we would likely miss Green Drakes, because they would all be eggs at that point and neither available to be captured in the sampler nor identifiable even if we were able to capture them. Exceptions to the one-year life cycle are some stoneflies such as the giant salmonfly, whose nymphs live several years before emerging and so will always be present no matter when we sample. Other exceptions could include small-bodied species that have adapted their life histories from a single brood per year to two broods per year in response to warmer water temperatures. Even in that case, we would still capture those individuals in mid-March sampling, as the two emergence periods would be spring and late summer or autumn.

Here is one important thing to note about our sample timing. If a given mayfly species is present in the sample in March and is also present at roughly the same number the following March, it means that the insects successfully reproduced. In other words, nymphs present in March of the first year survived and matured, the duns emerged, they molted into spinners, the spinners laid eggs, the eggs hatched, and the nymphs survived until the following March. This happened even if duns or spinners were never observed in large numbers at a given location on a given day of the fishing season at a given time of day. Further, even if a large hatch did not occur under the conditions to entice fish to rise on the surface, the nymphs provided food for fish feeding underwater for the 340–360 days they lived in the river.

Field sampling methods

Brett trained us in use of what is called a "Hess sampler", which is basically an open metal cylinder with mesh on the upstream side of the cylinder to allow water to flow into it and a net sticking out of the downstream side. The cylinder is placed on the stream bottom and pressed down roughly 3 inches into the bottom substrate, which can range from coarse sand to large cobbles at our Henry's Fork sites. We stir the stream bottom thoroughly to loosen the individual invertebrates, which float up into the water column and are carried by the current down into the net. We then pick up each individual rock within the sampler and scrape all invertebrates from the rock with a brush. These are also carried by the current down into the net. Everything in the net is poured into jars and preserved with alcohol. See the photo in Figure 2 of Brett and HFF's data manager Melissa Muradian using a Hess sampler.

In 2015, with little funding, a small staff, and no initial data, we collected three samples at each site. In 2016 and 2017 we collected five samples at each site to improve statistical power, and since then, we have collected six samples at each site each year. Although a larger number of samples is always better from a statistical standpoint, each sample costs almost $695 in staff time, travel, supplies, and lab costs. It takes four HFF staff and three of Brett's staff around nine hours to collect 30 samples each year at our five permanent sites. Six samples seem to provide a good balance between data quality and cost.

Laboratory analysis

The Hess sampler has a known area, in our case 0.1 square meter, so our sampling method is what is known as "quantitative" sampling. That means that we can estimate the number of insects per unit of area of river bottom and compare apples-to-apples estimates of abundance from year to year. Invertebrates in the Henry's Fork are so abundant that each sample contains several thousand individuals, prohibiting Brett from counting every single individual. Thus, he and his crew use what is known as "subsampling," which is a precisely defined method in which around 200 individuals from each sample (an average of around 7% in our case) are randomly selected from the full sample to be identified and counted. The method provides an accurate estimate of the subsampling fraction, so that results from the subsample can be scaled up and reported for the whole sample. As good as this method is, it can still result in very large or rare species being missed, so for the reporting of certain measures, Brett and his crew conduct what is called a "large and rare" search that is biased toward such species to make sure that the total number of species is reported as accurately as possible. We use data from the random subsampling for most of our analyses but use the large and rare value for total number of mayfly, stonefly and caddisfly species present in the sample. The numbers Brett reports for each sample are multiplied by 10 to obtain numbers per square meter of river bottom, and then we can use that to estimate total numbers of individuals over whole river reaches.

How did we analyze the data?

Invertebrate response variables

With hundreds of species observed over the 11 years of sampling, dozens of species that are of interest to anglers, and only 56 independent data points to work with, we had to pare hundreds of possible analyses down to a small number in order to learn as much as possible while minimizing the chance of obtaining spurious statistical relationships. We ultimately chose to analyze total invertebrate abundance, five so-called “community metrics” (EPT taxa richness, %EPT, HBI, %non-insects, and Shannon’s diversity index, discussed individually below in the results section), and abundance of Pale Morning Dun (PMD) mayflies, *Drunella* mayflies (Flavs *Drunella flavilinea* and Green Drakes *Drunella grandis* combined), and the Spotted Sedge caddis (Hydropsychidae). The community metrics are well established measures of the relative abundance and types of invertebrates that indicate the overall health and function of the river. They have been used all over the world for decades as indicators of water quality and aquatic ecosystem health. We chose to analyze PMDs and the Spotted Sedge caddis because those are the two hatches we hear most often from anglers that have declined substantially at Last Chance and the upper Ranch in recent years. We chose *Drunella* mayflies because they are iconic on the Henry’s Fork, important to anglers, and extremely sensitive to water quality and habitat degradation. *Drunella* mayflies and PMDs were collected at all sites in all years, and the Spotted Sedge caddis was collected at all sites in all years except Flat Rock in 2023, 2024 and 2025.

Environmental predictors

It is well established that the types and numbers of aquatic invertebrates in streams are dependent on streamflow, aquatic ecosystem productivity, fine sediment, and water temperature. We used eight variables measuring these four environmental characteristics that could potentially influence invertebrates in the Henry’s Fork.

1. **Annual streamflow (AnnQ)**. Overall water quality and habitat availability is expected to be better in years with higher streamflow.
2. **21-day minimum streamflow (Min Q).** Long-duration low flows limit the amount of habitat available in the stream channel, thereby potentially limiting the abundance of invertebrates.
3. **3-day maximum streamflow (MaxQ)** during the spring (prior to July 1). A higher springtime freshet is expected to remove fine sediment and have other positive effects on habitat quality.
4. **Streamflow variability (QCV),** as measured by the coefficient of variation in daily streamflow across the year. Some species prefer relatively stable flows, as would be observed in a groundwater-dominated stream or a constant-flow tailwater. On the other hand, high flow variability is likely to create a higher diversity of physical habitat, leading to higher species diversity.
5. **Suspended sediment concentration (SSC)** during summer/fall (July 1 – October 31). Sediment delivered to river reaches in the Henry’s Fork during this time period is likely to be trapped on the stream bottom by aquatic vegetation or deposited there during low-flow periods, thereby reducing habitat quality for sediment-intolerant species.
6. **Conductivity (Cond)** during the growing season (April 1 – October 31). Conductivity measures the concentration of soluble ions in the water. In the ranges we observe in the Henry’s Fork (80-150 µS/cm), conductivity is a surrogate for primary production potential, with higher values indicating higher productivity. We have never observed conductivity values anywhere close to the 1,000 µS/cm values that could indicate concentrations of salts or pollutants that could negatively affect freshwater fish and other organisms.
7. **7-day maximum water temperature (Temp)**. The literature suggests that maximum temperature limits presence of individual species, especially mayflies and stoneflies valued by anglers.

8. **Total phosphorus concentration (Phosphorus)** during summer/fall. Phosphorus is generally the limiting nutrient in aquatic systems, meaning that primary production increases when phosphorus concentrations increase. While phosphorus concentrations in the Henry's Fork are far below those that are toxic to aquatic insects, anglers have expressed concern that increased phosphorus concentrations due to expanded housing development and other human activities could negatively affect insect communities. Because of the expense of sampling phosphorus (currently $45 *per sample*) we have complete phosphorus data to match the invertebrate data only at Flat Rock, Last Chance, and Osborne Bridge, so only those sites were included in the phosphorus analysis (33 total data points).

These variables are calculated from raw data measured by HFF's water-quality sonde and stream gage network, our weekly water samples, and U.S. Geological Survey stream gages. The values were calculated over the year (or relevant portion of the year) immediately preceding annual mid-March sampling, to capture the habitat conditions present over the life span of the insects being collected.

While these fundamental characteristics are expected to influence aquatic invertebrates, we also tested the effect of location along the river, since other variables such as stream-bottom substrate type (sand, gravel, cobble, etc.), shading, stream width-to-depth ratio, and growing season length vary across locations and may not be completely represented by the streamflow and water-quality variables. The simplest potential effect of location is measured by **distance** downstream from the river's headwaters at Big Springs, according to the "River Continuum Concept" (RCC), a fundamental tenet of stream ecology. The RCC states that a river's ecosystem varies predictably from the headwaters downstream as the stream gets wider, the growing season gets longer, water temperatures increase, floodplain complexity increases, and the effects of watershed processes such as erosion and deposition accumulate over a larger area. The RCC predicts that with increased distance downstream, invertebrate abundance and species diversity should increase, while the fraction of species that are intolerant of fine sediment, habitat degradation, and warm temperatures should decrease.

However, we expect that the RCC may not apply very well to the Henry's Fork for three reasons. First, the river is groundwater fed, geologically very young, very wide relative to its depth right from the source at Big Springs, and generally disconnected from surface watershed processes because the watershed is flat and there are few tributaries connecting upland areas to the main river. Many of the tributaries are themselves groundwater fed, introducing large amounts of temperature-moderated flow at different points along the river. Thus, integration of additional watershed area with distance downstream does not predictably change the river's channel shape and temperature regime as it does in typical Rocky Mountain rivers. Second, the stream bottom substrate in the Henry's Fork is determined by local geology and not by distance along the continuum. Whereas the substrate in most rivers in the western U.S. predictably transitions from bedrock to boulder to cobble to gravel to sand/silt with distance downstream from headwaters, the stream bottom of the Henry's Fork randomly varies among coarse sand, bedrock, gravel, boulder/cobble, and cobble/gravel. For example, the substrate at the river's source is fine gravel and coarse sand, while there are several bedrock reaches on the lower river, downstream of St. Anthony. Third, the river continuum is interrupted by Island Park Dam, Ashton Dam, and diversion of much of the river's streamflow at and downstream of Chester Dam. These disruptions alter streamflow, temperature, and/or nutrient and sediment transport from what would be expected in an unregulated stream, thus altering otherwise predictable changes in the river as it flows downstream. Thus, we also tested the possibility that differences across **sites,** without regard to their distance along the river continuum**,** had the greatest impacts on aquatic insects and other invertebrates. Lastly, we tested whether the types and numbers of invertebrates present in the Henry's Fork show any systematic **trends** over our 11 years of investigation.

Statistical methods

If you do not water to read the details of the statistical methods, you can skip to the results section below. However, for full transparency and scientific rigor, the statistical modeling details are given here.

We used multi-model inference with Akaike's Information Criterion, corrected for sample size (AICc) to investigate dependence of the nine response variables on the predictors. We first averaged the replicate samples collected at a given site in a given year to obtain a single observation for each of the 56 unique site-year combinations. Because all sites experienced the same overall climate and water conditions in each year, the streamflow and water quality variables generally exhibited the same temporal pattern across years at each site (Figure 3), indicating the possibility of pseudo-replication within years across the sites. For example, in a year of high water supply, all sites had high values of mean annual flow. However, because the effects of water management in the regulated system differ greatly across river reaches depending on their position relative to the storage reservoirs and points of irrigation diversion, the relative magnitude of the flow variables with a given year of common water supply varied substantially across sites. The same thing was true of the water-quality variables. Further, the predictor variables were not highly correlated with one another. The highest correlation among the water quantity and water quality variables was $r = -0.79$ between streamflow coefficient of variation and 21-day minimum streamflow. The highest correlation between distance downstream from Big Springs and any of the other predictor variables was $r = 0.70$, with 7-day maximum water temperature. Thus, we considered the set of 56 observations across all site-year combinations as independent for the modeling of invertebrate responses to the streamflow, water-quality, and location variables.

Next, we assessed distribution of each of the response and predictor variables and applied appropriate transformations. The three streamflow magnitude variables, suspended sediment concentration, and total phosphorus were log_e-transformed. Flow coefficient of variation, 7-day maximum temperature, conductivity, and distance from headwaters needed no transformation. All of the invertebrate abundance metrics were strongly right-skewed and required log_e-transformation. A constant of 47 and 26, respectively, was added to the *Drunella* and Hydropsychidae abundance variables before applying the logarithm to further reduce skewness, and in the case of Hydropsychidae, to accommodate three 0 observations. The diversity index was left-skewed and best corrected with a 4^{th}-power transformation. The two fractional responses (percent EPT and percent non-insects) were logit-transformed, after conversion to decimals. No transformation was required for HBI or EPT taxa richness. Linear models (`lm` function in R) were used for the diversity, HBI, and EPT taxa responses. Generalized linear models (`glm` function in R) were used for the abundance variables, with Gaussian variance and log_e-link. Beta regression (`betareg` function in R) with logit link was used for the fractional responses. Residual diagnostics were performed to ensure that assumptions were met.

Multi-model inference was implemented with the `MuMIn` package, using the "all combinations" approach. After fitting all possible models, a subset was selected, consisting of all models with AICc less than that of the null model (intercept only) or all models with $\Delta AICc \leq 4$, whichever set was smaller. The AICc weights were recalculated on this subset and then used to calculate unconditional model-averaged coefficients and Wald's Z statistics (coefficient divided by standard error). Model-averaged fitted values were back-transformed if needed and used to calculate Efron's pseudo-R^2. This procedure was performed for each of three sets of potential predictor variables: 1) streamflow and water-quality variables only, 2) streamflow and water-quality variables plus distance from headwaters, and 3) streamflow and water-quality variables plus site as a categorical predictor (6 levels). For each of these three potential sets of predictors, we report the AICc value of the top model, Efron's pseudo-R^2 for the averaged model, and all averaged coefficients for which Wald's Z was 1.64 or greater in absolute value. In a hypothesis-testing context, these are coefficients that would be considered significant at $P < 0.1$. This same procedure was repeated for PMDs, with total phosphorus added as a potential predictor, but only at Flat Rock, Last Chance and Pinehaven (n = 33 site-year combinations), since those were the only sites with phosphorus data for every year of the study.

Mixed-effects models (`lme` function in R) were used to analyze trends in the response variables over time. To avoid pseudo-replication resulting from correlation in annual climate conditions across sites, site was used as a random grouping variable, with year as a continuous random effect within sites. Using the same response-variable transformations described above, two models were fit for each response: 1) a null model with no fixed effects, and a model with year as a fixed effect. The two models were compared using the likelihood ratio test, and the effect of year (temporal trend) was considered significant if $P < 0.05$. For the purposes of graphical representation, we also fit analogous trend models to each site on its own and considered the site-specific trend significant if $P < 0.0083$, using Bonferroni's correction applied to the family of 6 individual site tests at a family error rate of $\alpha = 0.05$.

Results

Overview

Overall, the results confirm scientifically what anglers already know, namely that each reach of the Henry's Fork is unique, and aquatic insects do not change predictably from headwaters downstream. The salmonfly hatch provides a good example of this. On rivers such as the Yellowstone and Madison, the hatch starts in the lower reaches of the river and then progresses up the river as water temperatures increase, whereas on the Henry's Fork, the hatch starts in ground-fed headwater areas, then jumps to the regulated mid-river reaches and lastly to the snowmelt-fed Fall River and reaches of the Henry's Fork immediately downstream. Accordingly, we found that sample site explained more variability in invertebrate communities than did location along the river continuum. Further, because each site has a unique combination of streamflow, water quality and habitat type, invertebrates showed very little dependence on streamflow or water quality. Over the past 11 years, seven of the nine metrics we analyzed were either roughly constant over time or improved. The observed trends suggest that invertebrate communities are shifting away from non-insects and toward mayflies, stoneflies and caddisflies and other insect species that are indicators of good habitat and water quality. The only two responses that showed significant declines at the watershed scale were total invertebrate abundance and PMD abundance. However, neither showed any correlation with streamflow or water-quality variables that could explain the declines and provide guidance for restoration or management actions that could reverse the declines. Details on the predictor variables and on each response follow.

Streamflow and water-quality characteristics

Before even looking at the insects themselves, the streamflow and water-quality data showed both expected changes along the river continuum and the disruptions to the continuum we anticipated due to the river's unique geology and the effects of dams and diversion (Figure 3).

The streamflow variables were divided by the 1978–2025 average streamflow at each site to allow apples-to-apples comparisons across sites at different locations in the watershed. This also allowed us to put the last 11 years in the context of the longer period of record, given that the river's natural streamflow was much greater in the 1970s–1990s that it has been since 2000. For example, you can see that over the last 11 years, annual streamflow at all locations was less than the long-term average (values on the graph less than 1) in almost all years of our study. It is also apparent that average annual streamflow within a given year was fairly constant from Flat Rock downstream to Marysville, since Island Park Dam changes only the timing of streamflow throughout the water year and not the total amount of annual flow. However, annual flow at St. Anthony was quite a bit different than at the other four locations because the fraction of the river's annual flow diverted for irrigation upstream differs substantially across years. Thus, the year-to-year variability in annual flow is much greater at St. Anthony. The apparent lower variability at Ashton Dam is just an artifact of having only three years of data there. The 3-day maximum flow largely reflects natural flow availability and was higher both in magnitude and variability at St. Anthony than at the other locations. Despite delivery of managed freshets at Island Park Dam during the spring of several years in the dataset, the magnitude and variability of maximum flows were lower at Last Chance and Osborne Bridge than elsewhere. Conversely, flow regulation at Island Park resulted in much higher variability in low flows—and generally lower minimums—at Last Chance and Osborne than at the other locations. Not surprisingly, because of adherence to a low-flow target at St. Anthony set to keep as much water in Island Park Reservoir as possible, minimum flows at St. Anthony were the lowest across all sites, with little year-to-year variability. Daily flow variability was highest—and had the highest variability from year-to-year—at Last Chance and Osborne Bridge. Taken together, the flow variables clearly show the effects of storage and delivery at Island Park Dam and diversion between Ashton Dam and St. Anthony, illustrate that flow regimes do not follow predictable changes along the river continuum, and indicate that our sample sites vary substantially in flow characteristics despite experiencing the same overall water supply each year.

Maximum water temperatures generally increased with distance downstream as would be expected, with the exception of higher temperatures at Osborne Bridge as a result of the naturally wide, shallow, unshaded river reach through Harriman State Park. In this case, the disruption to the river continuum comes from the natural geology of the river, not from Island Park Dam. While conductivity generally increased from headwaters to St. Anthony as would be expected, conductivity at Osborne Bridge was higher than expected because of higher summertime water temperatures there. That said, conductivity varied relatively little across sites and was very low in comparison to most other trout streams in our region because the volcanic rocks in the Henry's Fork watershed contain very low concentrations of soluble ions. Suspended sediment concentrations also generally increased with distance downstream from headwaters as would be expected, with the exception of much higher sediment concentrations at Last Chance due to export of fine sediment from Island Park Reservoir during the summer and fall. Most of that fine material is either trapped or consumed between Last Chance and Osborne, where suspended sediment concentrations are about what we would expect at that location along the river in absence of Island Park Dam. Lastly, phosphorus concentrations were higher at Last Chance than at the other locations but showed relatively little variability across locations, keeping in mind that complete 11-year records of phosphorus were available only at Flat Rock, Last Chance, and Osborne Bridge.

Invertebrate abundance

We measured abundance as the number of individuals per square meter (m^2) of stream bottom. Abundances ranged from 12,390 individuals/m^2 at Last Chance in 2022 to 86,540 at Osborne Bridge in 2015. The average over all sites and years was 35,647 individuals/m^2, or in more familiar units, around 3,313 individuals per square foot. To give you some idea of just how many individual insects (and other invertebrates) that is, the river at Last Chance is about 300 feet wide, so one mile (5,280 linear feet) of river there has an area of 300×5280 = 1.58 million square feet. Average abundance at Last Chance is around 2,800 individuals per square foot, so one mile of river in the vicinity of our Last Chance sampling location contains about 4.4 billion individual invertebrates. As a more specific example, in 2025 there were around 1.7 billion Pale Morning Duns and 225 million Flavs per mile at Last Chance.

You may be asking, "Is 35,647 individuals/m^2 high or low?" Comparable data are hard to come by, because few organizations or institutions collect data that are this extensive and rigorous. However, the Bighorn Alliance and the Upper Missouri Watershed Alliance each conduct annual macroinvertebrate monitoring using similar methods as HFF. The average abundance in springtime samples taken on the Bighorn River from 2021 to 2023 is 30,000 individuals/m^2, comparable to the Henry's Fork. The average abundance reported from the upper Missouri River between 2015 and 2023 is somewhat lower, at around 12,500 individuals/m^2, but some of the samples were taken in the summer and fall, when abundances are likely to be lower. Average abundance was 21,697 individuals/m^2 in the Henry's Fork at Last Chance and Osborne Bridge in the spring of 1993, after the 1992 sediment event at Island Park Dam. Average abundance at Coffee Pot (comparable to our current Flat Rock site) in the spring of 1993—affected by upstream land use and the 1988 Yellowstone fires but not the reservoir sediment event—was around 12,000 individuals/m^2. In his work around the western U.S., Brett Marshall has observed abundances upwards of 100,000 individuals/m^2, and in fact we have observed abundances close to 100,000 individuals/m^2 in some years on the South Fork Snake River. In these cases, however, the samples are dominated by midges and/or non-insects to the point where 100,000 individuals in total may contain far fewer mayflies, stoneflies, and caddisflies than present in 35,000 individuals on the Henry's Fork.

Further insight into abundance on the Henry's Fork is provided by the upper left panel in Figure 4, which shows abundance values as a function of distance down the river continuum, with the sites indicated along the top of the graph. We found no change in abundance along the river continuum and no average difference across our sampling sites. In addition, we found no statistical evidence for dependence of abundance on any of the streamflow or water-quality variables (Table 2), and the statistical models explained only 9% of total variability in abundance. That means that abundance of invertebrates in the Henry's Fork is essentially constant across our sampling sites and although variable from year to year, does not vary in response to any of the streamflow or water quality variables.

We rarely hear that hatches have declined in river reaches such as Warm River to Ashton or Ora to Vernon, which have the same overall invertebrate abundances as places like Last Chance, where we consistently hear that hatches have declined. If 35,000 individuals/m^2 are sufficient to maintain good hatches, healthy fish, stable and abundant trout populations, and good fishing conditions in the river reaches downstream of Warm River, any perceived decline in the fishing experience in the Ranch must be due to some factor other than the number of invertebrates. We discuss some of those near the end of the report.

One particular speculation we hear is that lack of aquatic invertebrates (i.e., food) is the cause of low trout populations between Island Park Dam and Riverside. If that were the case, then we would expect to see low trout populations in all reaches of the river, given that they all have the same average invertebrate abundance as Last Chance. Further, we observed no correlation between invertebrate abundance at Last Chance or Osborne Bridge and the trout population in Box Canyon (Figure 5), which is an indicator for the population from Island Park Dam to Riverside.

That said, we have observed a statistically significant ($P = 0.003$) decreasing trend of around 4% per year over the past 11 years at the watershed scale (Table 3, Figure 6), although none of the site-level trends were significant. As will be discussed in subsequent sections, the overall decline in abundance is accompanied by improvements in three of the community metrics, suggesting that decreased abundance is likely due to replacement of non-EPT species and those that are indicative of degraded habitat conditions with mayflies, stoneflies, and caddisflies that are indicative of better conditions.

Diversity of Invertebrates

Diversity of an ecological community is a measure both of how many different types of organisms are present and how the total number of individuals is distributed across those different types. Higher diversity is indicative of high availability of a variety of different habitat types and of good function of the ecological processes that maintain those habitat types. In stream ecosystems, this means that streamflow regimes, riparian conditions, stream substrate scour and deposition, and nutrient availability are sufficient to maintain good numbers of a variety of different invertebrate types. Further, high diversity indicates a high degree of resilience to changes in any of these characteristics; if one element of the stream ecosystem changes and no longer favors a particular species, there is always another one present that will benefit from the change. We measured the diversity of the invertebrate community with Shannon's diversity index, a very common measure used in ecology and other scientific disciplines. Shannon's index ranges from 0, in the extreme case in which the community consists of only one species, to the natural logarithm of the total number of different species present, in the case that each species is equally represented. So, for example, if 20 species are present, the theoretical maximum diversity would be the logarithm of 20, which is 3. In our samples, the number of species present ranged from about 20 to 40, with an average close to 30. That means that the maximum diversity possible in the Henry's Fork ranges from around 3 to 3.7, if those 20–40 species were equally represented.

Diversity ranged from 1.7 at Flat Rock in 2023 to 3.1 at St. Anthony in 2016 and averaged 2.7 across all sites and all years. These numbers indicate very diverse invertebrate communities overall, with values at St. Anthony close to the maximum possible for the given numbers of species present. Shannon's diversity is not reported directly in publicly available data from the upper Missouri and Bighorn, but the total number of species reported on the Bighorn River is comparable to those in our Henry's Fork samples, around 20-40. In the spring of 1993, Shannon's diversity averaged 2.4 at Last Chance and Osborne Bridge and 2.2 at Coffee Pot, just a little lower than what we have observed. Over all samples of invertebrates collected on the Henry's Fork between 1993 and 2007, Shannon's index averaged 2.1, quite a bit lower than our modern average of 2.7. Unlike abundance, diversity increased significantly with distance downstream (Figure 4 and Table 2), as would be expected from the RCC. However, the best statistical model included site rather than distance from Big Springs as a predictor and also included suspended sediment concentration, with higher diversity being associated with lower suspended sediment concentrations, after accounting for differences across sites. While we observed a significant change in diversity across sites, we have observed only a marginal change in diversity over time (P =

0.051; Figure 6 and Table 3), indicating robust, stable, and resilient invertebrate communities, especially in the lower watershed.

Number of Mayfly, Stonefly, and Caddisfly Species (EPTT)

These three insect orders (**E**phemeroptera, **P**lecoptera, and **T**ricoptera, respectively) are not only the most important to anglers but are also highly indicative of water and habitat quality. Thus, a standard metric for assessing water quality is the so-called EPT Taxa richness, EPTT for short. The term "taxa" is used instead of species to acknowledge the fact that sometimes, it is not possible to distinguish individual insect larvae ("nymphs") down to the species level, so that higher taxonomic levels such as genus or even family might be used in the calculation. Regardless, you can think of it as the number of different species of mayflies, stoneflies, and caddisflies present. In our samples, EPTT ranged from 8 at Osborne Bridge in 2020 to 17 at Marysville in 2023, with an average of 13 over all sites and years. Generally, EPTT values of 10 or more (or roughly 30% of all species present) are considered good, and our values fall in that range, similar to those observed on the Bighorn River. The number of EPT taxa increased slightly with distance downstream from Big Springs (Figure 4 and Table 2), but the greatest variability in EPTT was explained simply by difference across sites (Table 2). Marysville consistently had the highest EPTT, while Ashton Dam had the lowest, keeping in mind that we only have three years of data at Ashton Dam. However, even the best statistical model for EPTT accounted for only 32% of the observed variability, and EPTT did not depend on any of the streamflow or water quality variables. We observed a statistically significant increase ($P = 0.002$; Table 3) in EPTT over the past 11 years at the watershed scale and a significant increase at Marysville (Figure 6). At the watershed scale, the number of EPT taxa per site increased by an average of 2.5 species over the past 11 years. The combination of stable community diversity and increasing EPTT indicates that non-EPT taxa are being replaced by mayflies, stoneflies, and caddisflies.

Hilsenhoff Bioitic Index (HBI)

The HBI score is a widely used index of water quality, specifically degradation due to organic pollution such as untreated wastewater, fertilizers, petrochemicals, and pesticides. The score is based on tolerance of each type of invertebrate to such pollution. The most sensitive species, such as Flavs and the giant salmonfly, have a tolerance score of 0, while the species most tolerant of water pollution have a score of 10, for example many aquatic worm species. Most mayfly, stonefly and caddisfly species have tolerance values in the range of 1–4. For example, tolerance scores for Pale Morning Duns, *Tricorythodes* mayflies (tricos) and *Brachycentrus* caddis (for example, the "Mother's Day Caddis") have tolerance scores of 2, 4, and 1, respectively. The HBI is simply the average of species-level tolerance scores across all individuals present in the sample, where the average is weighted by the number of individuals of each species. This produces a score between 0 and 10, where 0 is indicative of the best possible water quality, and 10 is indicative of the worst water quality. Specifically, larger numbers of intolerant species (most of the mayflies, stoneflies, and caddisflies) will produce a lower score. The numeric scores are interpreted qualitatively according to a commonly used scale:

- 0–3.75: Excellent water quality; no apparent organic pollution
- 3.76–4.25: Very good water quality; slight organic pollution possible
- 4.26–5.00: Good water quality; some organic pollution apparent
- 5.01–5.75: Fair water quality; fairly significant organic pollution
- 5.76–6.50: Fairly poor water quality; significant organic pollution

- 6.51–7.25: Poor water quality; very significant organic pollution
- 7.26–10: Very poor water quality; severe organic pollution

In our samples, HBI ranged (worst to best) from 5.9 at St. Anthony in 2015 to 2.4 at Flat Rock in 2016 and averaged 4.0 over all sites and all years. So, our average score falls into the "very good" range, indicating possible slight organic pollution. The average HBI over all invertebrate samples collected on the Henry's Fork between 1993 and 2007 was 4.2, indicating no substantial change over the past 30 years. For comparison, the average HBI reported for the Missouri River is 5.7 and that for the Bighorn River is around 6.0, with most sites in most years having HBI scores greater than 5, falling into the "fair" range, two qualitative categories below the Henry's Fork. As predicted by the River Continuum Concept, we observed a significant increase in HBI score (decrease in water quality) from headwaters to St. Anthony (Figure 4 and Table 2), although again, site was a better predictor of HBI than distance downstream. This is primarily because HBI was a little higher than expected (worse water quality) at Osborne Bridge and a little lower (better water quality) at Marysville. After accounting for the differences across site, HBI was lower (better water quality) in years following higher annual streamflow and warmer temperatures. Over the past 11 years, the HBI has improved significantly ($P = 0.044$; Table 3) at the watershed scale and at Osborne Bridge (Figure 6). At the watershed scale, HBI improved from 4.4 in 2015 ("good" category) to 3.9 in 2025 ("very good" category), indicating overall improvement in water and habitat quality.

Percent Non-insects

This is simply a measure of the fraction of individuals in the sample that are not insects. These include worms, leeches, and snails. Although many of these species provide fish food and are relatively intolerant of pollution, their presence is often associated with high amounts of fine sediment and other types of physical habitat degradation. Percent non-insects ranged from 7.8% at St. Anthony in 2024 to 38.7% at St. Anthony in 2015 and averaged 21% across all sites and years. We found no difference in percent non-insects along the river continuum or across sites (Figure 4 and Table 2) and no significant dependence on any of the streamflow or water-quality predictors (Table 2). At the watershed scale, percent non-insects showed a marginal decrease over the study ($P = 0.055$; Table 3), as significant decreasing trends at Osborne Bridge and Marysville were outweighed by a significant increasing trend at Flat Rock.

Percent Mayflies, Stoneflies, and Caddisflies (%EPT)

This is a measure of the fraction of individuals present in the sample that are mayflies, stoneflies, and caddisflies and, along with HBI, is a standard measure of water quality. Values greater than 30% are considered good. In our samples, %EPT ranged from 22.2% at St. Anthony in 2016 to 74.9% at Flat Rock in 2016, with an average of 53% over all sites and years. This is similar to the average of 51% reported for the upper Missouri but quite a bit higher than the average of around 30% on the Bighorn. The average over all data we have from the Henry's Fork from 1993 to 2007 is 49%, pretty close to its current average. We observed a significant decrease in %EPT with distance downstream, as would be expected by the River Continuum Concept (Figure 4). However, as with HBI, variability across individual sites was greater than predicted by the river continuum, again driven primarily by higher values than expected at Marysville. Also similar to HBI, we found that after accounting for the site differences, %EPT was higher when temperatures were warmer (Table 2), and %EPT has improved significantly over the past 11 years (P = 0.006; Table 3) at the watershed scale, although none of the site-level trends were significant (Figure 6). On average, %EPT has increased from 45% to 59% over the course of the study.

Given an increase in EPT Taxa and %EPT and a marginal decrease in percent non-insects, it is apparent that the decrease in overall abundance has resulted mostly from decreases in non-insects and in insects other than mayflies, stoneflies, and caddisflies (midges, for example). The result is an overall improvement in the community composition, potentially reflecting improved habitat conditions and ecological function, especially at Osborne and Marysville.

Pale Morning Duns

Abundance of PMDs ranged from 365 individuals/m^2 at St. Anthony in 2017 to 30,511 at Flat Rock in 2016, with an average of 7,697 individuals/m^2 over all sites and years. Another way of looking at PMD abundance is that it has ranged from 2% to 61% of the total invertebrate community, with an average of 21%. That is, over all sites and all years, over 20% of all individual invertebrates present on the stream bottom are PMDs. Data from the Bighorn River show that PMDs are typically less than 4% of all individuals. We have not dug into the details of Henry's Fork datasets from the 1990s and 2000s enough to pull out estimates of PMD abundance. However, in our modern data, PMD abundance shows a systematic decrease with distance downstream, as we would expect from the River Continuum Concept (Figure 7). Unlike in the statistical models for EPTT, HBI, and %EPT, the river continuum dependence provided a better predictive model of PMD abundance than individual differences across the six sites. After accounting for dependence on distance downstream from Big Springs, none of the streamflow or water quality variables had any explanatory power (Table 2). This is especially noteworthy because 7-day maximum temperatures have generally been in the upper 60s to mid-70s (Fahrenheit) over the course of our study, and the literature suggests a maximum of tolerance of around 65 degrees for PMDs. Thus, PMDs are persisting in the Henry's Fork at much higher temperatures than suggested by other studies, and we saw no strong relationship between PMD abundance and temperature, even in models without location as a predictor. That could be because the temperatures recorded at our sonde locations are not fully indicative of localized cooler water refuges associated with groundwater inputs, which we know occur throughout the river. That said, PMD abundance has systematically increased across the watershed at a rate of around 7% per year for the past 11 years (P = 0.004; Table 3), although none of the site-level trends were significant (Figure 8). PMD abundance at Last Chance showed relatively low year-to-year variability and a trend line that was essentially flat (Figure 8).

When the PMD analysis was restricted only to Flat Rock, Last Chance, and Osborne Bridge, and total phosphorus was included as a predictor, the best statistical model again included distance downstream as a predictor, but also included 7-day maximum water temperature as a predictor (Table 4). However, the dependence on water temperature was positive, indicating higher PMD abundance following years of higher water temperature. PMD abundance did not depend on total phosphorus.

To dig a little deeper into PMD abundance, scatterplots of abundance vs. all predictors illustrate the relatively low correlations with streamflow and water-quality variables (Figure 9). The strongest correlation was $r = -0.42$ with flow variability, followed by $r = -0.39$ with 7-day maximum water temperature and $r = 0.39$ with 21-day minimum streamflow. These correlations are all driven by the very high abundance of PMDs at Flat Rock (blue points) relative to all other locations. Because Flat Rock has by far the lowest flow variability, lowest maximum temperature, and highest minimum flow of all sites (Figure 3), these apparent correlations with streamflow and temperature actually reflect correlation with location, not the streamflow and water-quality variables themselves. Further evidence is that the correlation between PMD abundance and location along the river continuum is $r = -0.59$, much greater than any of these other correlations. Once the effect of location is accounted for, correlations with most of the streamflow and water-quality become weaker, and some even reverse sign. For example, PMD abundance is highest at locations with cooler water temperatures and lowest at locations with warmer water temperatures, as we would expect, but within a given site, PMD abundance shows little dependence on water temperature or any other water-quality or streamflow variable from year to year.

Drunella (Flav + Green Drake) abundance

Drunella abundance ranged from 51 individuals/m^2 at St. Anthony in 2017 to 4,822 individuals/m^2 at Last Chance in 2020 and averaged 1,311 individuals/m^2 over all sites and all years. This is an average of 4% of all invertebrates present at any given time and location on the Henry's Fork. Again as expected, *Drunella* abundance decreases with distance down the river (Figure 6), although numbers at Last Chance are much higher than expected based solely on distance. The best statistical predictor of *Drunella* abundance was simply differences across the individual sites, with no environmental predictor adding any explanatory power (Table 2). *Drunella* abundance shows no significant trend one way or the other across the watershed; a significant decrease at Flat Rock was offset by a significant increase at Osborne Bridge (Figure 8).

Hydropsychidae Caddis (Spotted Sedge) abundance

Spotted Sedge abundance ranged from 0 at Flat Rock in 2023, 2024 and 2025 to 8,206 individuals/m^2 at Marysville in 2021 and averaged 1,502 individuals/m^2 across all sites and years. On average this is about the same as abundance of Flavs and Green Drakes. However, Spotted Sedge numbers were much more variable across locations, being far more abundant at Last Chance and Marysville than anywhere else (Figure 7). After accounting for this substantial variability across sites, Spotted Sedge numbers were higher following years of higher suspended sediment concentrations (Table 2). Spotted sedge abundance has increased significantly by around 3% per year at the watershed scale (P = 0.043; Table 4) and has increased significantly at Ora Bridge and St. Anthony (Figure 8).

What do these results mean?

While it is obvious that this dataset—combining rigorous invertebrate sampling with detailed water-quality and streamflow monitoring—provides an unprecedented view of aquatic ecosystem function in a unique river system, the results will no doubt disappoint many anglers, who would like to see a "smoking gun" that points to an easily identifiable and hopefully rectifiable reason for the apparent decline in hatches at Last Chance and the upper Ranch. The reality is that even a dataset this rich cannot capture the level of detail in insect life histories and behavior to explain why a particular hatch occurs at a particular location at a particular time of day. And remember where we started; to have rising fish, you need fish, and we know with certainty that trout numbers at Last Chance and in the Ranch are around 50% to 75% lower than they were in the 1970s through 1990s due to lower water supply. But, here are some take-home messages from the insect part of the equation.

The Henry's Fork Invertebrate community is robust and stable

By any measure, the aquatic invertebrate community up and down the river is abundant, diverse, and indicative of good to excellent water and habitat quality. Over half of the individuals are mayflies, stoneflies, and caddisflies, and that percentage has been improving over the past decade, along with the number of mayfly, stonefly, and caddisfly species present. While total abundance is decreasing at the watershed scale, the decrease is accompanied by increases in %EPT and EPTT and a marginal decrease in %non-insects, suggesting that non-insects and non-EPT taxa are responsible for the majority of the losses. Further, improvement in %EPT, EPTT and HBI indicate overall improvement in water and habitat quality. All metrics on the Henry's Fork are as good or better than on the two other popular trout rivers from which we have comparable data, and conditions over the past 11 years on the Henry's Fork are at least as good as they were when measured between 1993 and 2007 and in most cases are better. Upstream of Island Park Reservoir, improvements are likely related to recovery from sediment originating with the 1988 Yellowstone fires, improvements in grazing management on Henry's Lake Flat implemented in the 1990s, and more stable outflow from Henry's Lake over the past decade. Immediately downstream of Island Park Dam, improvements are due to recovery from the 1992 sediment event, some occurring incidental to routine water management and others occurring because of intentional release of freshet flows to remove fine sediment from the stream bottom. In the lower watershed, improvements are likely due to more careful streamflow management designed first and foremost to save water in Island Park Reservoir but that also have the benefits of reducing variability in summertime streamflow downstream of St. Anthony and allowing a more natural hydrograph there during spring runoff.

Fine sediment and water pollution are not limiting invertebrates

Suspended sediment concentration was a significant statistical predictor of only two of our response variables, and one of those relationships was positive—more sediment was associated with higher Spotted Sedge abundance. This is because that species feeds on fine particles of organic matter suspended in the water column. We observed a negative dependence of community diversity on suspended sediment concentration but otherwise found no evidence that suspended sediment was associated with any other community metrics. Further, the HBI scores indicate little to no effects of organic pollution overall, although we saw some evidence of effects of organic pollution at Osborne and St. Anthony during the first few years of the study, which occurred during or immediately after the extended drought of 2013–2016 and prior to implementation of precision water management actions in 2018 that have reduced summertime flow variability at St. Anthony. We also observed evidence of organic pollution at St. Anthony in 2022, after the very dry year of 2021. Indeed, water supply was a significant predictor of HBI, with lower values (better water quality) following years of good water supply. We would expect this observation, since higher streamflow provides more dilution of any pollutants that might be present. However, while HBI varies from year to year according to water supply, the facts that the average HBI is within the "very good" range and that HBI shows a significant improvement over time provide strong evidence that water pollution is not affecting the quality of the Henry's Fork aquatic invertebrate communities.

Trends at Flat Rock are concerning

Despite our observations that conditions over the whole watershed appear to be stable if not improving, Flat Rock is an exception. While most trends at Flat Rock are not statistically significant, they are generally opposite of those at the watershed scale. Specifically, %non-insects, EPTT, % EPT, and HBI are all improving or marginally improving at the watershed scale, but all show the opposite at Flat Rock (Figure 6). *Drunella* abundance is stable at the watershed scale while decreasing significantly at Flat Rock, and decline in PMD abundance at Flat Rock is a major contributor to the watershed-scale decline (Figure 8). However, even with these watershed-scale trends, water temperature, suspended sediment concentrations, and phosphorus concentrations at Flat Rock are consistently lower than at any of the other sites, and streamflow at Flat Rock is the least regulated by storage reservoirs and irrigation diversion of all sites. So, the invertebrate trends we are seeing at Flat Rock are puzzling and concerning. These declines could be occurring because Flat Rock is the most pristine of our sites and hence has the most to lose in a changing climate. In other words, a couple of degrees of warming may not make that much of a difference at St. Anthony, where summertime water temperatures are already well into the 70s and have been for decades. But at Flat Rock, where temperatures have historically stayed in the 50s and 60s all summer, a couple of degrees is proportionally a much greater increase. Further, we know that the location in the Henry's Fork watershed that has seen the greatest decrease in streamflow per unit of precipitation is the upper Henry's Fork. While annual water supply over the whole watershed (including Fall River and Teton River) has decreased by 15% since 2000, water supply upstream of Island Park Dam has decreased by 22%. We observed statistical evidence that HBI scores are higher (worse) when streamflow is lower, and this would have the greatest proportional effect at Flat Rock.

No invertebrate trends are apparent at Last Chance

None of the nine response variables we analyzed show any systematic trend over the past decade one way or the other at Last Chance, and that includes PMD abundance, which is of greatest concern to anglers. Our data show that average PMD abundance at Last Chance is far lower than that at Flat Rock, so it is possible that PMDs at Last Chance were formerly as abundant as they still are at Flat Rock and declined substantially prior to the initiation of this study. We will do some digging into the data archives here to see if we can find some comparable numbers for PMDs from the 1970s, 1980s and 1990s, but at least over the past 11 years, we have no evidence of the "catastrophic" decline in PMD hatches at Last Chance and the upper Ranch that we have been hearing a lot about over the past few years. Certainly there is year-to-year variability, but the point is that the invertebrate community at Last Chance has not changed systematically over the past 11years. Further, the year-to-year variability in PMD abundance at Last Chance we have observed over the past 11 years does not generally correspond with angler experience. More on that in the next section.

Insects on the stream bottom are not the same as a hatch

This research is based on invertebrates that we sample during their nymph stage and is not designed to predict when or where a given aquatic insect hatch will occur. We focus on nymphs as they are present on the stream bottom in March because they provide the greatest amount of information about water and habitat quality. Further, an aquatic insect's role in the stream ecosystem occurs primarily when it is in the nymph stage. After all, most of the species anglers are interested in spend 350 days in the river as nymphs and only two weeks or less, and in most cases only a few days, as winged adults. Yes, those few days provide the core of the Henry's Fork fishing experience, but it's the other 350–360 days that reflect the overall health and productivity of the river and how it might respond to changes in water quality, streamflow, and climate.

To be a little more specific, the five years in our study with the highest PMD abundance at Last Chance were 2025, 2017, 2020, 2018, and 2023 (Figure 10). Of these, by far the year with the best perceived hatches was 2020—and even most long-time anglers said that 2020 had the best hatches in decades, comparable to what was commonplace in the 1970s and 1980s. Yet, PMD abundance was over 50% greater in 2025, generally considered the worst fishing year ever on the Ranch. By far the two years with lowest PMD abundance at Last Chance were 2022 and 2024 (Figure 10). Based on angler comments, PMD hatches in 2024 were widely considered (at the time) to be the worst ever at Last Chance and the upper Ranch, yet there were fewer PMDs in our samples in 2022, when we heard relatively little about poor hatches. Further, we generally heard the same negative comments about PMD hatches in 2023 as we heard in 2024 and 2025, yet PMD abundance in 2023 was over three times what it was in 2024 and the highest on record in 2025. Thus, there is little to no relationship between PMD abundance and angler satisfaction with PMD hatches.

However, angler perception of hatches does appear to be correlated with fish numbers and secondarily with turbidity. While we do not have statistically rigorous data on angler satisfaction, anglers do not hesitate to register their concerns and complaints with us when fishing does not live up to expectations. Over the course of the 11 years represented in this study, we universally heard that fishing was the worst on record (at the time) in 2016, PMDs had disappeared in 2023, and hatches were even worse in 2024. By 2025, anglers had little chance to even fish Last Chance and the upper Ranch due to turbidity, as overall water quality was even worse than it was in 2016 and the trout population was equally low. Essentially, 2016, 2023, 2024, and 2025 were all "worst on record," each being a little worse than its predecessors. While high turbidity could explain angler dissatisfaction in 2016, 2024, and 2025, turbidity was high in 2020 and 2021, when dry-fly fishing was good. On the other hand, turbidity was low in 2023, when fishing and hatches were considered poor.

Thus, the quality of the dry-fly fishing experience has little relationship with abundance of particular mayfly species, a somewhat stronger relationship with water quality, and probably the strongest relationship with trout numbers. However, we think temperature holds the biggest key to explaining changes in fishing experience that long-time anglers have observed at Last Chance and the upper Ranch.

What about temperature?

Our water-quality sonde data show a statistically increasing trend in water temperatures over the past 11 years, and that trend applies over the whole watershed. Average summer water temperatures have increased by around 2 degrees F over the past 11 years, while 7-day maximum temperatures have increased by 1 degree F. As discussed above, almost all of the 7-day maximum temperatures we observed were above the maximum tolerance of PMDs reported in the literature and also above the maximum tolerance of other mayflies such as blue-winged olives (*Baetis*) and Flavs that are abundant and widespread in the Henry's Fork. So, either the temperature tolerances observed in studies conducted in other river systems do not apply to the Henry's Fork or the numerous groundwater inputs to all reaches of the Henry's Fork maintain locally cooler areas in the river than where our water-quality sondes are located.

In models that did not contain a location variable, water temperature appeared as a significant predictor three times, and all were consistent with overall expectations. Shannon's diversity index increased with water temperature, as would be expected based on the River Continuum Concept; warmer sites are expected to occur lower in the watershed, where habitat diversity is greatest. The HBI was higher, indicating lower water quality, where temperatures were higher. Abundance of *Drunella* was higher at lower water temperatures, as would be expected for this temperature-sensitive species. However, after accounting for variability across location in the watershed, these temperature dependencies either disappeared (diversity and *Drunella*) or reversed sign (HBI). Temperature appeared as a significant predictor for %EPT after location was included as a predictor, and the sign was also the opposite as would be expected in that case; higher temperatures resulted in higher %EPT.

These results are consistent with the detailed results on PMD abundance discussed above. For the most part, our results reflect expected patterns at the watershed scale: invertebrate communities are generally "better" as indicated by standard metrics and by presence of sensitive EPT species such as PMDs and *Drunella* at locations where temperatures are cooler. However, after accounting for differences in location, invertebrate communities show little response to temperature from year to year within a given site and when they do, the response is the opposite of what is expected. It is possible that large shifts in community structure and abundance of temperature-sensitive species occurred prior to our study, and our 11 years of record are not sufficient to see much more than random year-to-year variability within the "new normal". The exception may be Flat Rock, which is only now undergoing a substantial shift in species composition because it is the coldest of our six sites, and we started this study just in time to document the change there.

Regardless of large community-level shifts, it is very likely that increased water temperatures are affecting the life cycle of the insects that are present. It is well known that emergence behavior of aquatic insects is strongly determined by water temperature, and even small water temperature changes could result in large changes in emergence timing, both time of year and time of day. Further, our Harriman State Park temperature study shows that water temperature regimes (both the daily average and the variability around that average) vary greatly with location around the Ranch. As atmospheric temperatures warm, the difference between "warm" locations and locally "cool" locations will be greater than in the past, with hatches of specific insects being much more specific to location and time of day. The more localized a particular insect emergence is, the less likely it is that large numbers of the same insect will appear across large reaches of the river at the same time. This occurrence is exactly what produces the hatches for which the Henry's Fork is famous.

As a final observation about temperature, we have been integrating all that we know about invertebrates, temperature and streamflow downstream of Island Park Dam and have put together what we think is probably the most likely explanation for changes in hatches, especially PMDs, between Last Chance and Pinehaven. First, we know that outflow temperatures from Island Park Dam are lower when the reservoir stays full. Second, we know that water temperatures between Last Chance and Pinehaven are more resistant to warming from solar radiation and to cooling from the localized effects of cool groundwater inputs when outflow is high. Third, we know from our own angling experience and from the angling community in general that PMDs (and a few other mayflies) produce the most prolific hatches when water temperatures are relatively cool and relatively constant. That is why spring creeks and large reservoir tailwaters are generally most well known for their PMD hatches. In the Ranch, this would occur when Island Park Reservoir stays full but outflow is high. Over the past several decades, this condition is not physically possible, because high outflow leads to high reservoir drawdown, which leads to warmer outflow temperatures. Keeping the reservoir as full as possible requires low outflow, which is then susceptible to rapid warming once it hits Last Chance, especially because air temperatures are warmer now. However, back in the 1970s, 1980s, and 1990s, reservoir *inflows* were so high that it was possible to maintain high reservoir levels and high outflows at the same time. This would have produced a situation in which cool reservoir water remained cool all the way through the Ranch (not to mention the fact that air temperatures were lower), thus producing PMD hatches in large numbers at predictable times of year and times of day, all the way through the reach from Box Canyon to Pinehaven at the same time.

HFF actions moving forward

- Continue to monitor invertebrates, water quality, and streamflow to add to the 11-year dataset presented in this report.
- Review our old hard-copy reports for invertebrate data collected prior to this study and incorporate those into formal analysis as much as possible.
- Use newly completed water temperature and reservoir models to estimate what temperature regimes might have looked like in the 1970s-1990s, as a potential explanation for changes in both community structure and emergence behavior.
- Continue the collaborative precision water management program with Fremont-Madison Irrigation District and other partners that has saved over 23,000 ac-ft of water per year in Island Park Reservoir, increased winter outflow by 90 cfs, and resulted in small but measurable improvements in summertime water quality downstream.
- Contingent upon funding availability, pursue infrastructure and habitat improvements in Island Park Reservoir and on the river downstream to decrease turbidity and water temperatures.
- Develop and implement research in the future that will specifically investigate "hatches" as their own phenomenon, not necessarily reflective of the invertebrate community as a whole. We are implementing a pilot study along these lines in the spring of 2026, in collaboration with The Salmonfly Project, a sister non-profit conservation organization.

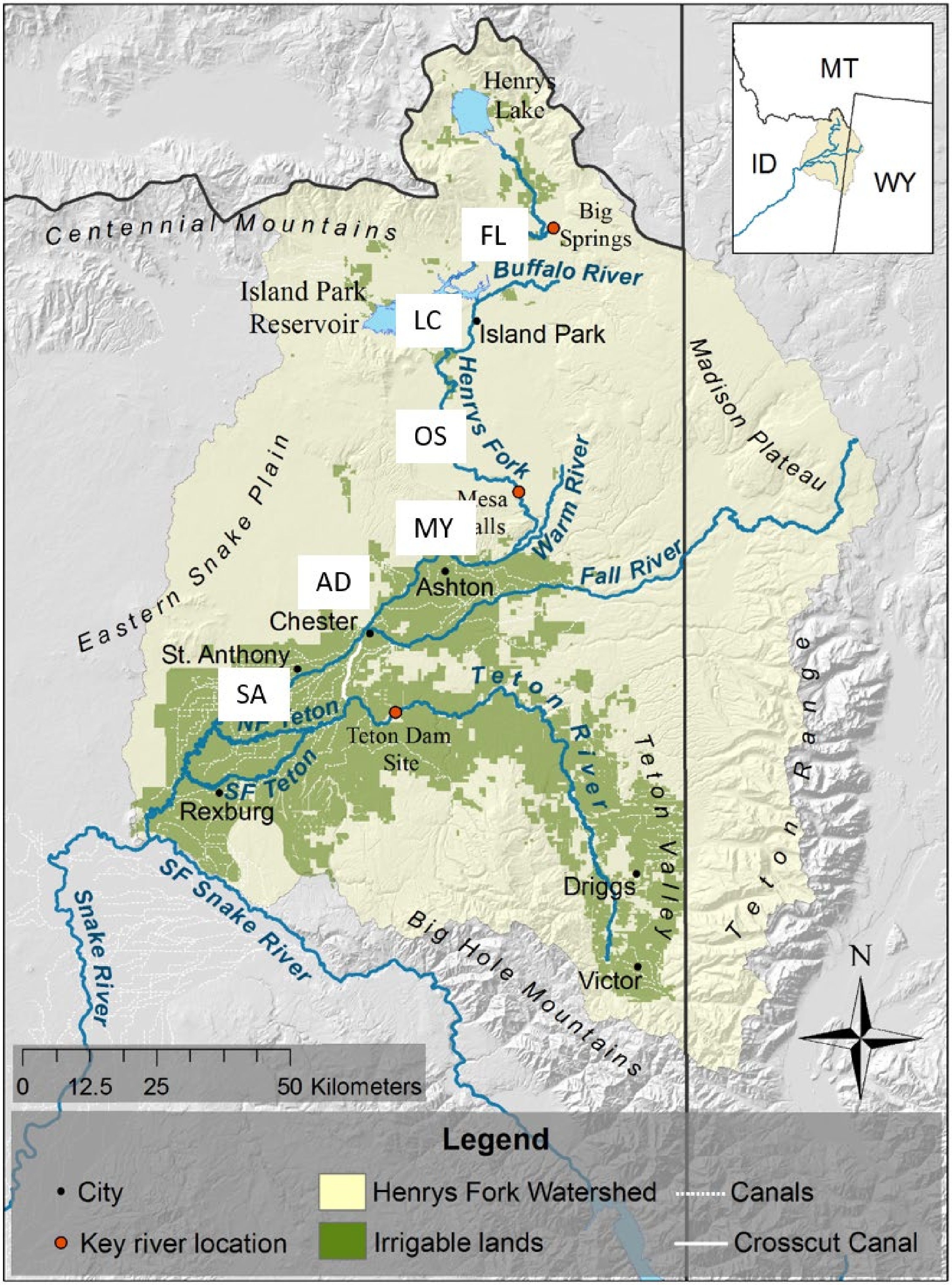

Figure 1. *Map of invertebrate sampling locations.*

***Figure 2**. Brett Marshall and Melissa Muradian use the Hess sampler to collect invertebrates at our Marysville site.*

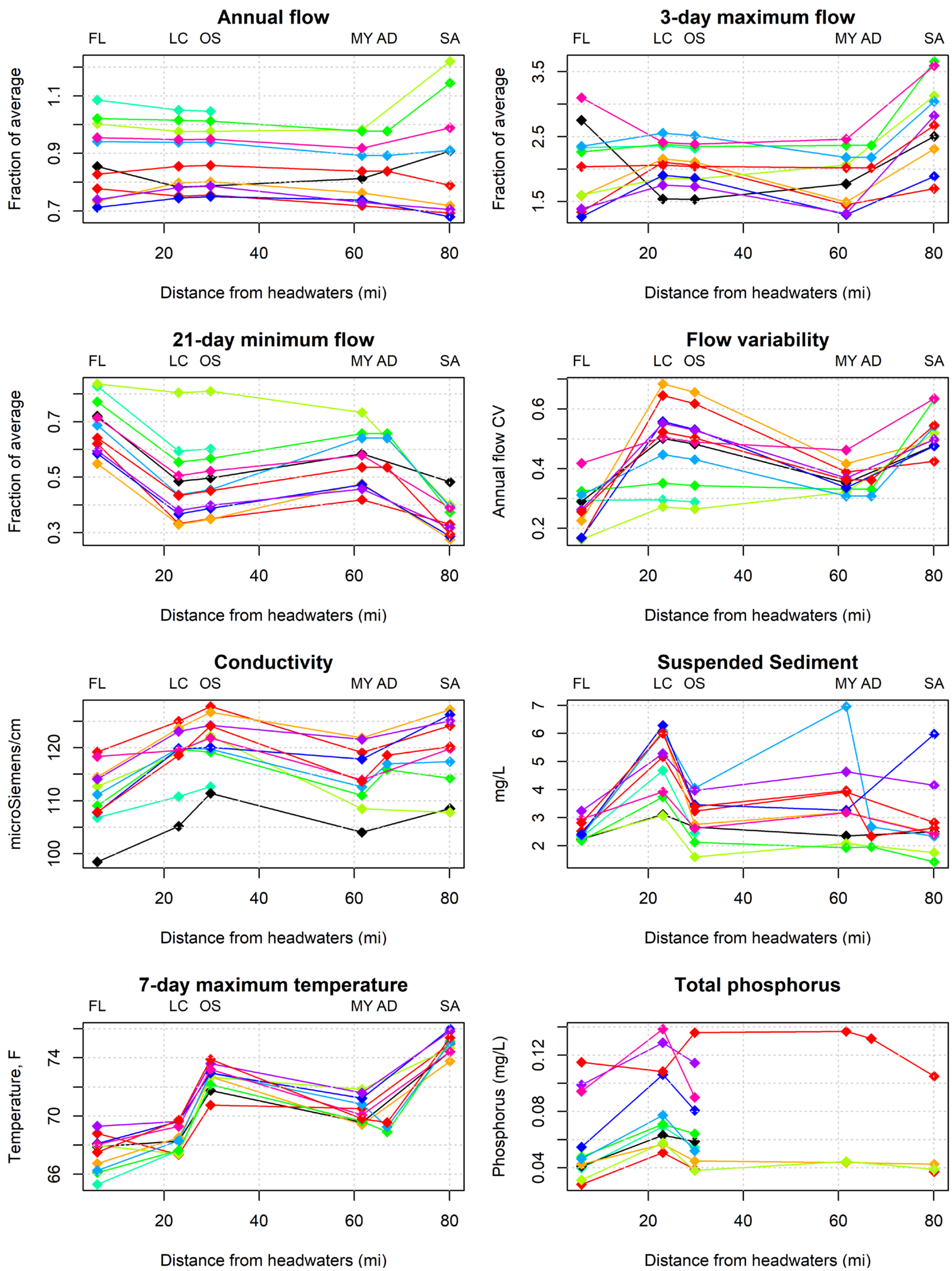

***Figure 3**. Habitat data for each site and year, plotted vs. distance downstream from Big Springs. Each sampling year is indicated by a unique color, and all points collected in that year are connected by lines.*

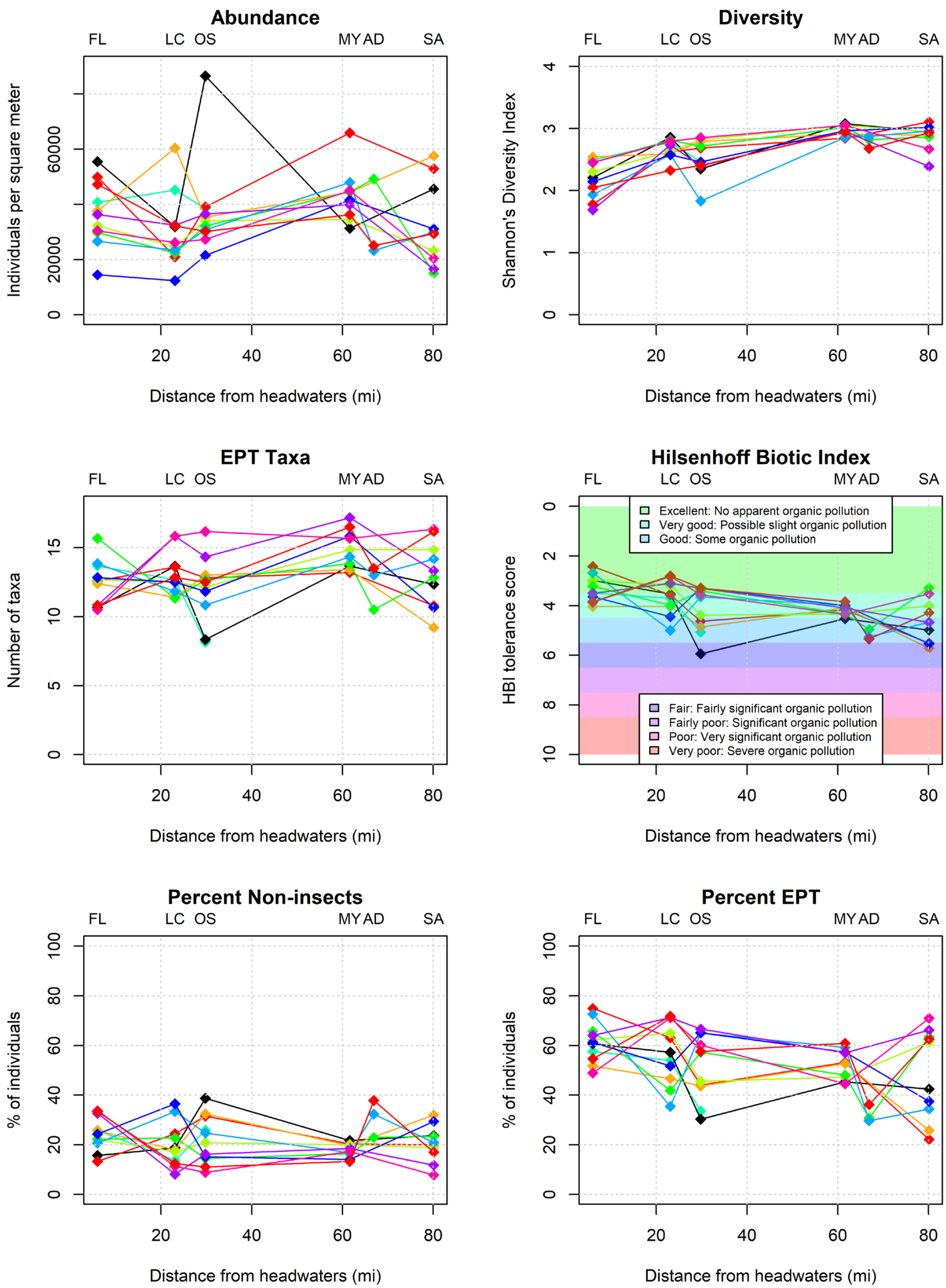

Figure 4*. Invertebrate community metrics for each site and year, plotted against distance downstream from Big Springs. As in Figure 3, colors correspond to each of the 11 sampling years.*

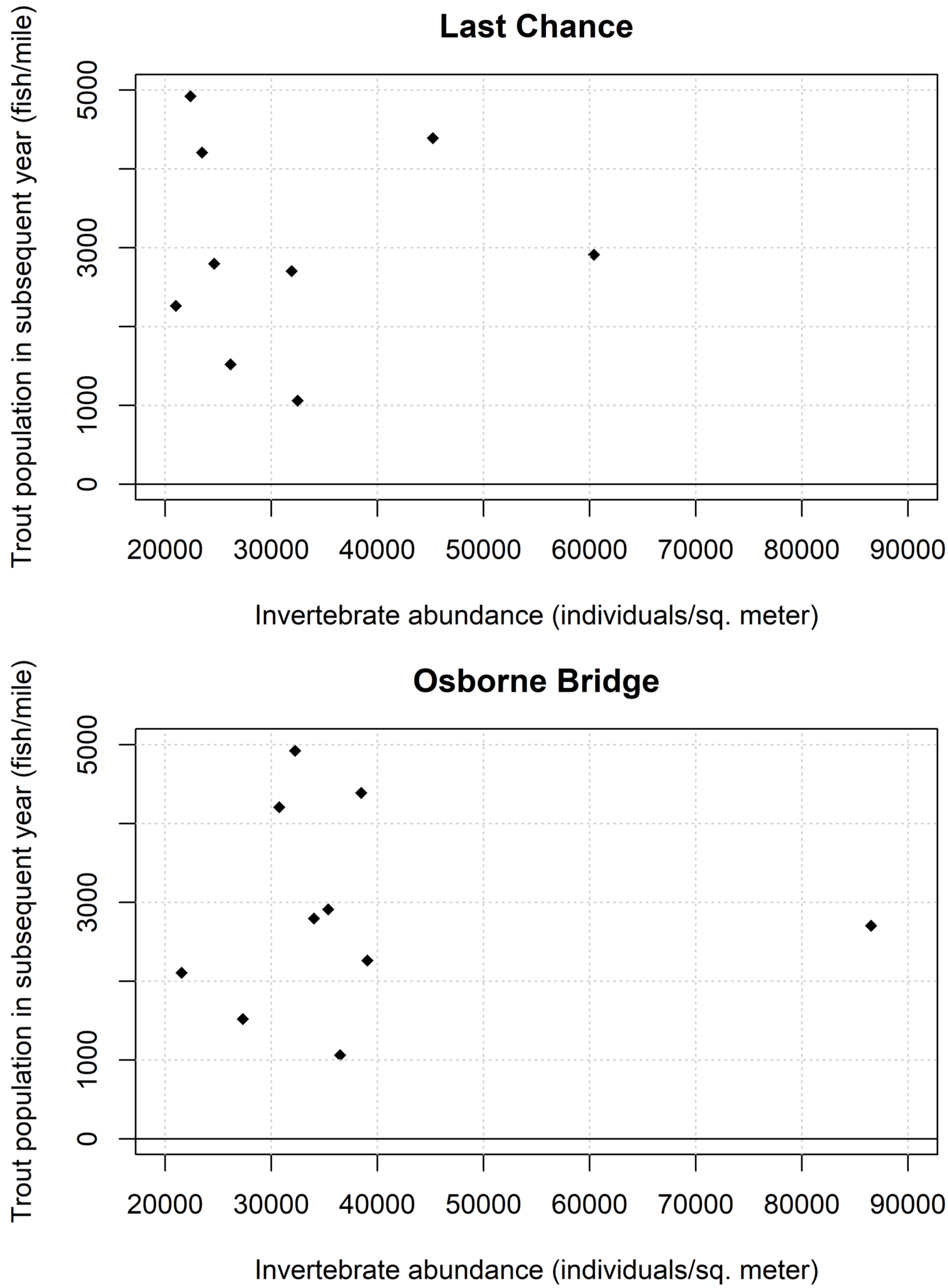

Figure 5*. Box Canyon trout population plotted as a function of total invertebrate abundance at Last Chance and Osborne Bridge. Correlations are* $r = 0.13$ *at Last Chance and* $r = 0.0015$ *at Osborne.*

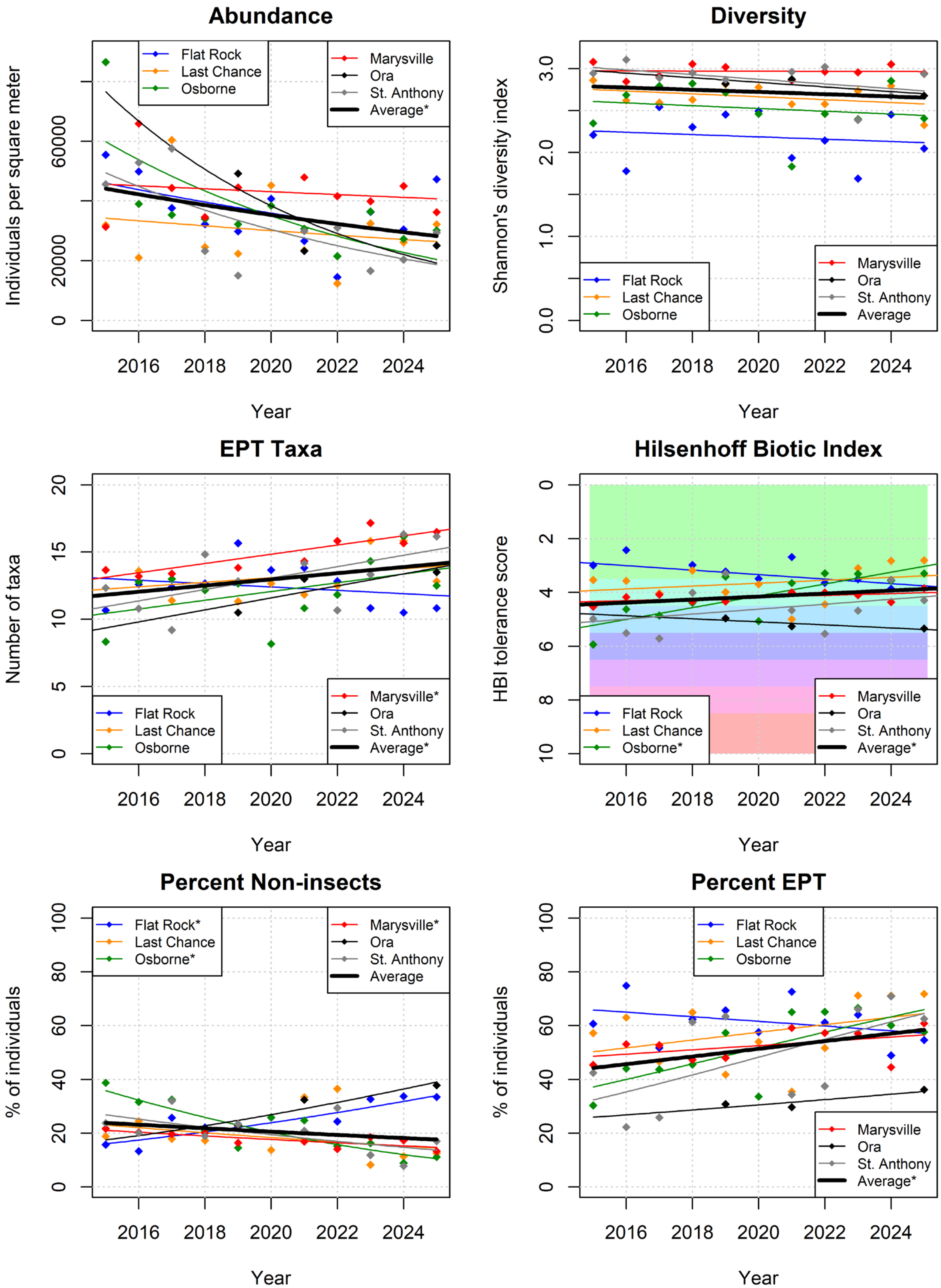

Figure 6. *Time series of invertebrate community metrics for the five sites with 2015–2025 data. Asterisks indicate statistically significant trends over time. Colors in this graph indicate the six sampling sites.*

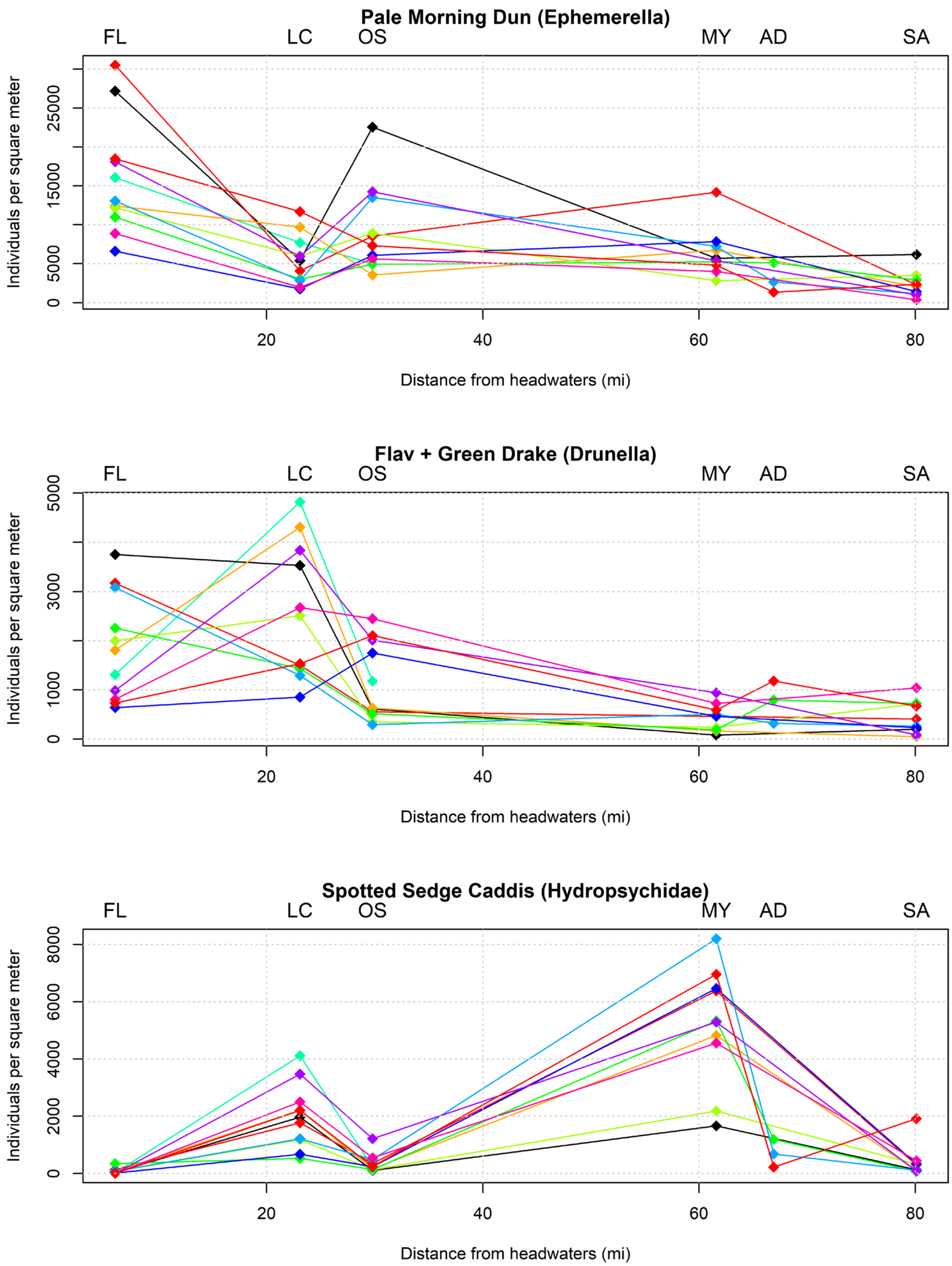

Figure 7*. Abundance of PMDs, Drunella, and Spotted Sedge for all sites and years, plotted by distance downstream of Big Springs. As in Figures 3 and 4, colors correspond to each of the 11 sampling years.*

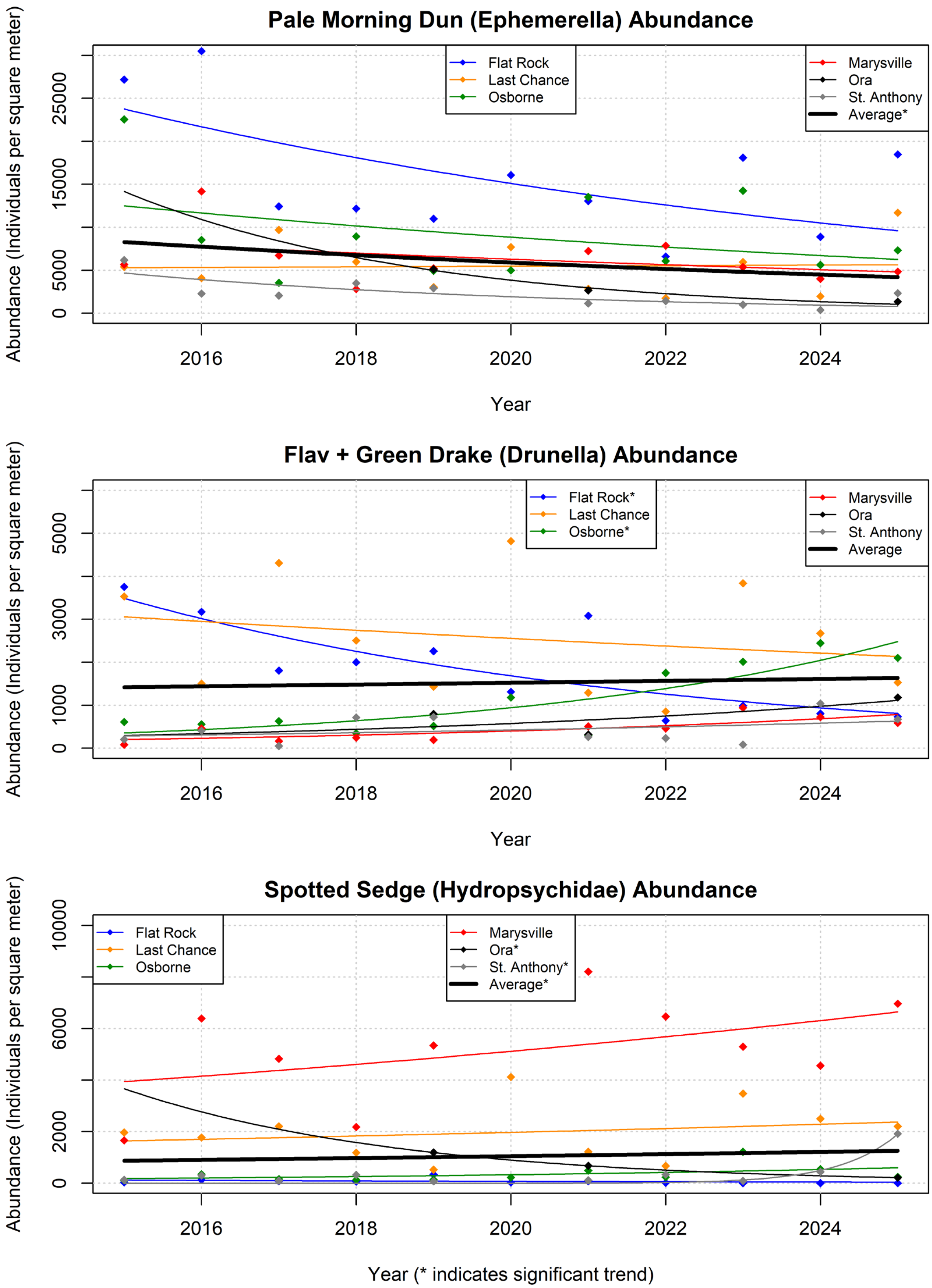

Figure 8*. Time series of PMDs, Drunella, and Spotted Sedge for the five sites with 2015–2025 data. Asterisks indicate statistically significant trends over time. Colors correspond to sampling sites.*

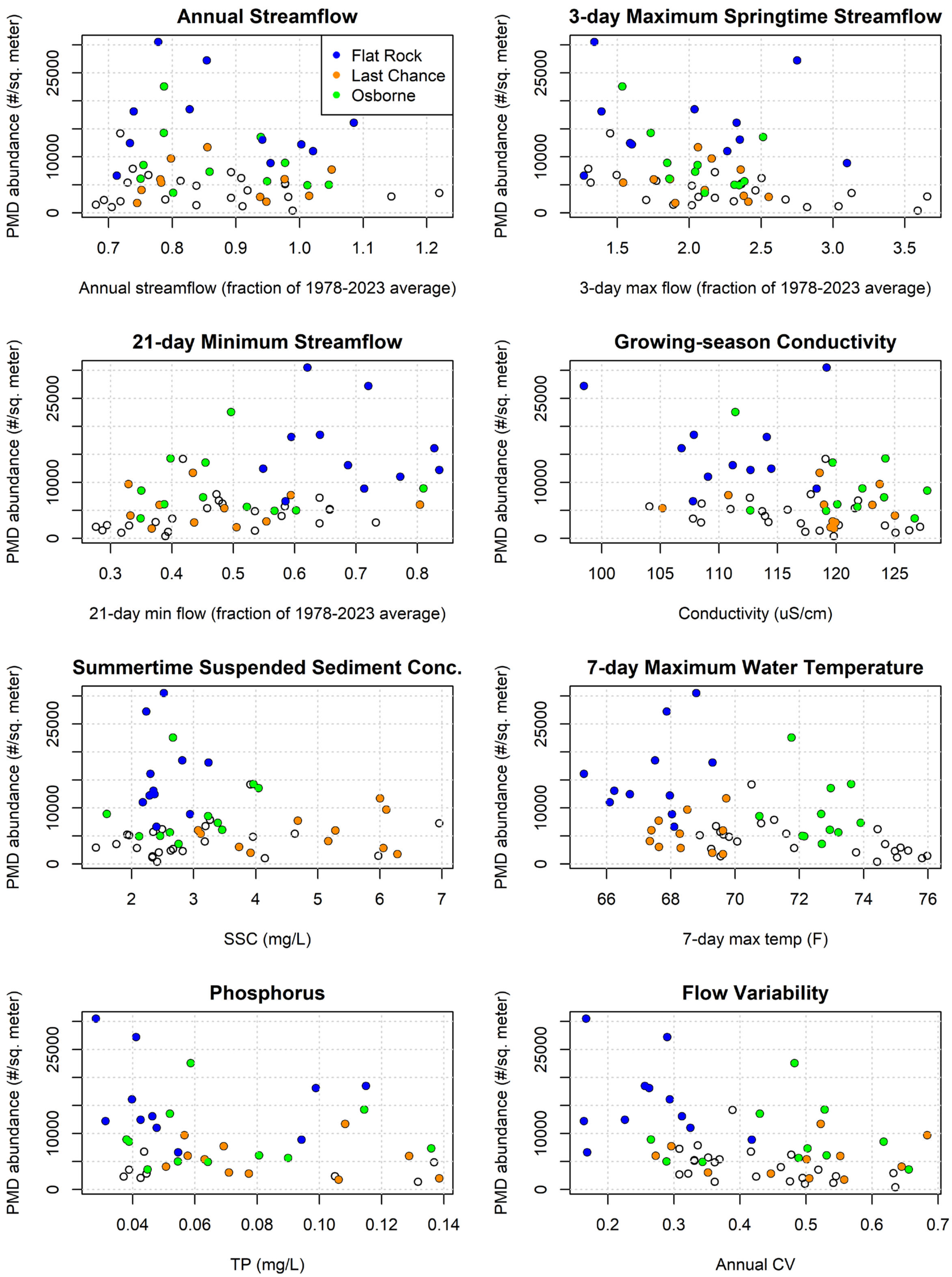

***Figure 9**. Scatter plots of Pale Morning Dun abundance vs. the eight streamflow and water-quality variables, with samples collected at Flat Rock, Last Chance and Osborne Bridge identified by color.*

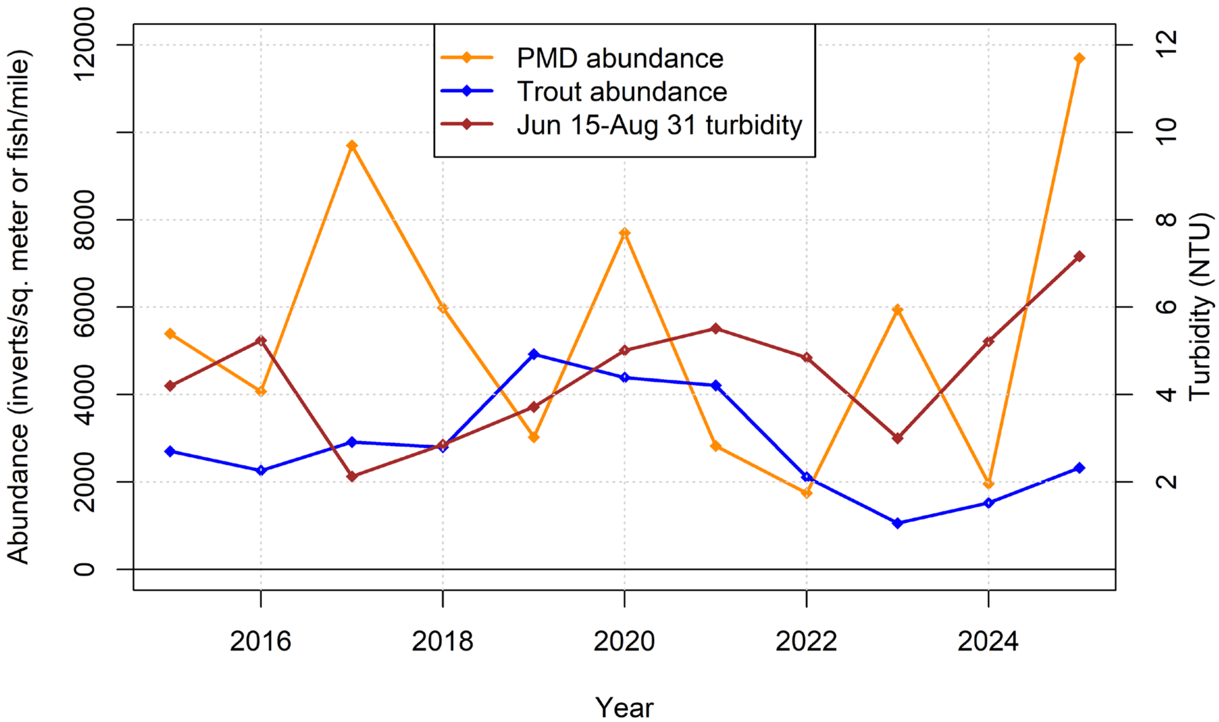

***Figure 10**. Time series plot of PMD abundance, Box Canyon trout abundance, and average June 15 – August 31 turbidity at Island Park Dam.*

***Table 1**. Aquatic invertebrate sampling locations on the Henry's Fork.*

Map ID	Name	Colloquial location description	Elevation (feet)	Dist. from Big Springs (miles)	Years sampled	Streamflow location	HFF water quality sonde(s)
FL	Flat Rock	Flat Rock Club	6388	6.0	2015-2025	Coffee Pot	Flat Rock
LC	Last Chance	Btw. LC boat ramp and Angler's Lodge	6171	23.1	2015-2025	Box Canyon (Island Park + Buffalo)	Island Park East + Buffalo
OS	Osborne	Btw. Hwy. 20 bridge and stock bridge	6112	29.8	2015-2025	Pinehaven	Pinehaven
MY	Marysville	Btw. Jumpoff Cyn. and Hwy. 20 bridge	5167	61.6	2015-2019 2021-2025	Ashton	Marysville
AD	Ashton Dam	Ora boat ramp	5107	66.9	2019, 2021, 2025	Ashton	Ashton Dam
SA	St. Anthony	Riverside Cemetery	4922	80.1	2015-2019 2021-2025	Trestle (St. A. minus diversions)	St. Anthony

***Table 2**. Summary of AIC model selection results. Lower AICc values indicate better predictive models. The model with lowest AICc for each response variable is indicated with shading. Wald's Z is listed only for AICc-averaged model coefficients with Z > 1.64 (equivalent to statistical significance P < 0.1). Z for the SITE variable is the largest in absolute value among the five fitted coefficients. Q = streamlow.*

Response	Location term	Top-model AICc	Ave. model R^2	Average model coefficient Wald's Z (estimate/SE) if absolute value > 1.64 (equiv. to P < 0.1)								
				Distance	SITE	AnnQ	MinQ	MaxQ	QCV	SSC	Temp	Cond
Abundance	None	1226.5	0.090	NA	NA							
	Distance	1226.5	0.093		NA							
	Site	1226.5	0.090	NA								
Diversity	None	504.8	0.10	NA	NA						2.60	
	Distance	456.5	0.65	8.58	NA						-2.68	
	Site	450.8	0.70	NA	8.93					-2.93		
EPT Taxa	None	241.6	0.00	NA	NA							
	Distance	240.9	0.07	1.65	NA							
	Site	238.7	0.32	NA								
HBI	None	133.5	0.18	NA	NA						1.74	
	Distance	120.0	0.39	4.34	NA							
	Site	115.8	0.55	NA	5.45	-2.59					-2.69	
%Non-insects	None	-126.1	0.00	NA	NA							
	Distance	-126.1	0.00		NA							
	Site	-126.1	0.00	NA								
%EPT	None	-66.4	0.00	NA	NA							
	Distance	-73.0	0.20	-2.85	NA							
	Site	-73.9	0.38	NA	-4.60						2.43	
PMD	None	868.0	0.35	NA	NA	-3.01	3.10					
	Distance	854.7	0.53	-2.89	NA							
	Site	857.0	0.57	NA	-2.96							
Drunella	None	683.4	0.29	NA	NA						-3.42	
	Distance	675.0	0.42	-3.32	NA							
	Site	670.3	0.51	NA	-2.67							
Spotted Sedge	None	746.2	0.35	NA	NA			-1.86		4.67		
	Distance	703.3	0.74	5.39	NA	2.11		-3.49		7.23	-3.69	
	Site	666.3	0.87	NA	3.24					4.69		

***Table 3**. Summary of watershed-scale trend analysis. Trends were considered significant if P < 0.05.*

Response	11-year trend	P-value
Abundance	**Decrease** of 4% per year	0.003
Diversity	Marginal decrease	0.051
EPT Taxa	**Increase** of 2.5 additional species over 11 years	0.002
HBI	Decrease of 0.6 over 11 years (indicates **increase** in water quality)	0.044
%Non-insects	Marginal decrease	0.055
%EPT	**Increase** of 1.3 percentage points per year	0.006
PMD	**Decrease** of 7% per year	0.004
Drunella	None	0.430
Spotted Sedge	**Increase** of 3% per year	0.043

***Table 4**. Summary of AIC model selection results for PMD abundance, restricted to Flat Rock, Last Chance, and Osborne Bridge, with total phosphorus included as a potential predictor. Lower AICc values indicate better predictive models. The model with lowest AICc for each response variable is indicated with shading. Wald's Z is listed only for AICc-averaged model coefficients with Z > 1.64 (equivalent to statistical significance P < 0.1). Z for the SITE variable is the largest in absolute value among the five fitted coefficients.*

	Location	Top-model	Ave.	Average model coefficient Wald's Z (estimate/SE) if absolute value > 1.64 (equiv. to P < 0.1)									
Response	term	AICc	model R^2	Distance	SITE	AnnQ	MinQ	MaxQ	QCV	SSC	Temp	Cond	Phosphsorus
	None	673.5	0.39	NA	NA								
PMD	Distance	668.4	0.51	-3.32	NA						2.46		
	Site	670.4	0.53	NA	-2.16								