Monitoring Aquatic Insect Densities and Environmental Factors on the Teton River, Idaho – 2023 Report

Authors:
Jackson H. Birrell[1*], James I. Frakes[1].

Affiliations and contributions:
[*] Lead contributor
[1] The Salmonfly Project, 123 W. Central Ave, Missoula, Montana 59801; conservation@ salmonflyproject.org

Summary:
Aquatic insects have declined across the Western United States, yet little is known about why and to what degree. This knowledge gap prevents action to protect and manage wild fisheries, biodiversity, and fishing opportunities at both local and regional levels. To begin filling this gap, we established a long-term insect and habitat monitoring program on one of the most economically and ecologically important fisheries in western US: the Teton River in north eastern Idaho. Historically, few systematic efforts have been made to monitor aquatic insect populations on the Teton River. The primary goals of our project are to annually: i) track the population densities of target insect species that represent the most abundant and important insects on the River and ii) analyze how important environmental factors, including temperature, flow, and fine sediment levels are correlated with insect densities and potential declines over time. In this report, we summarize results of the first year of sampling (2023), highlighting the distribution patterns of each target species and how environmental factors are correlated with species densities. Overall, our data provides little evidence of severe aquatic insect declines, yet shows that additional dewatering and sedimentation may make some reaches of the river uninhabitable to select species in the future. Indeed, while water temperatures on the Teton River are well-within tolerance limits of most resident taxa, the River is estimated to be ~ 30% dewatered. Distributions and densities of skwala, golden stoneflies and green drakes are also negatively correlated with fine sediment levels throughout the River, suggesting that abundant fine sediment may be suppressing populations at some sites. We emphasize the necessity of continual monitoring for tracking any future shifts in insect assemblages and environmental conditions and for identifying conservation priorities. Results from our data power analyses, however, suggest that maintaining a relatively large number of samples-per-site (i.e., ≥ 6), along with insect collection methods that reduce sources of randomness and bias, will be necessary for accurately detecting subtle insect declines, going-forward.

Data collected September, 2023; Report submitted September, 2024

Introduction:

Ecologically sensitive aquatic insects – including species of stoneflies, mayflies, and caddisflies - have declined across the Western United States, with recent examples highlighted by both scientists and anglers (e.g., DeWalt et al. 2005; Stagliano 2013; Giersch et al. 2017; Birrell et al. 2019; Stepanian et al. 2020, Sautner 2023; Bonavist 2023). Such declines are concerning because of the combined ecological and economic importance of aquatic insects: they feed game fish, like trout, and insectivorous birds, transfer nutrients to riparian ecosystems, and support local economies by creating renowned angling opportunities (Allan & Castillo 2007; Macadam & Stockam 2015). *Despite their importance and the widespread acknowledgement of their declines, little is known about where, why, and to what degree aquatic insects are declining in western rivers* (e.g., Birrell et al., 2020). Indeed, aquatic insects are rarely included in most federal and state conservation plans because of a lack of information about species' local vulnerabilities (e.g., Montana FWP 2015; USDA 2011). This knowledge gap hinders effective management of wild fisheries and the biodiversity, economies, and fishing opportunities they support. Generating new information to solve the insect conservation challenge will thus require implementing widespread, systematic monitoring programs to track the status, trends, and drivers of insect declines and identify conservation needs and solutions.

Implementing such programs on economically and ecologically important fisheries should be most heavily prioritized, especially those will little data on local insect populations, such as the Teton River, Idaho. The Teton River is well-known for its wild scenery, abundant trout, and diverse insect hatches, which draw anglers from around the world. anglers on the Teton, however, have also reported changes in the strength and timing of insect hatches, along with reduced flows and warmer water temperatures, which they suspect may contribute to reduced insect health and biodiversity. Unfortunately, historic scientific data on the Teton River's insects are sparse, and, until now, there haven't been any systematic insect monitoring programs to validate claims or determine why insects populations may be shifting. The current conservation status of aquatic insects, thus, remains widely unknown on the Teton River, highlighting the need to establish an annual insect monitoring program.

To help fill this gap, we initiated an aquatic insect and habitat monitoring program across eight sites on the Teton River, from South Bates Bridge to Teton Dam. The goals of the program are to annually: i) track the population densities of eleven target insect species that represent the most abundant and important insects the River and ii) measure how temperature, river flows, and fine sediment levels are correlated with insect densities and potential declines over time. A quantitative, target species approach was employed to reduce costs and increase feasibility of long-term funding – a major barrier for other monitoring programs that accrue large taxonomy expenses from monitoring all species.

Here, we report results for the first year of sampling, performed in September, 2023. Data outputs here, and in future years, will be distributed among local managers and stakeholders to help identify at-risk insects and potential threats that should be mitigated. Ultimately, these efforts will facilitate the proactive conservation of the Teton River and the local economies it supports.

Methods:

Target species:

We selected eleven target taxa of aquatic insects to monitor on the Teton River: giant salmonflies (stonefly; *Pteronarcys californica*), least salmonflies (stonefly; *Pteronarcella badia*); golden stoneflies (stonefly; *Hesperoperla pacifica*), nocturnal stoneflies (stonefly; *Classenia sabulosa*), skwala (stonefly; *Skwala americana*), mahogany duns (mayfly; *Paraleptoplebia spp.*), green drakes (mayfly; *Drunella grandis*), march browns (mayfly; Rhithrogena spp.), mother's day caddis (caddisfly; *Brachycentridae.*), and spotted sedges (caddisfly; Hydropsychidae). These species (or genera for *Paraleptophlebia* and *Rhithrogena* and family for Brachycentridae and Hydropsychidae; hereafter species) were thought to be among the most dominant insect hatches on the Teton and are thus of primary importance for both the ecology of the River and its recreational, cultural, and economic value. Target species also have readily-visible diagnostic features, allowing for rapid identification and enumeration. Higher taxonomic resolution for mahogany duns, march browns, Mother's Day caddis, and spotted sedges was not feasible because the necessary diagnostic features are not easily visible with the naked eye. Choosing important yet readily identifiable species was a primary goal of the project to ensure a useful, yet low-cost monitoring program. Nymphs of other important hatches on the Teton, including pale morning duns (mayfly; Ephemerella spp.) and yellow sallies (stonefly; Isoperla spp., Sweltsa spp., and others), could not be monitored because the nymphs were too small to catch or had not hatched from their eggs at the time of collection. Due to effects of seasonality on nymphal densities, data collection in future years will occur on approximately the same dates (within seven days) as this first year of sampling.

Monitoring sites:

We collected insect target species and measured fine sediment and embeddedness levels at eight sites along the length of the Teton between South Bates Bridge and Teton Dam, Idaho from September 20-29, 2023 (Fig. 1). Sites were riffled reaches with cobble-dominated substrate. Sites further upriver were not sampled because they were far deeper, slower flowing, and siltier, making quantifiable insect sampling difficult with the method we used. Sites were spread at roughly even intervals across the river.

Insect and environmental measurements

To measure the densities of target species, fine sediment levels, and embeddedness at each site, we took two samples along three transects of a 30m section of riffle at each site (six samples per site). At each transect, sampling locations were first chosen by walking a randomly chosen number of paces from shore and placing a 0.25m^2 rebar square on the substrate once the predetermined step-count had been met. All locations were chosen at water depths between 15-50cm, and sampling was performed from the downstream to upstream ends of the sampling area.

Fine sediment levels were measured with a sampling frame method similar to that of Bunte & Abt (2001). Measurements were made by placing a 0.5m diameter steel hoop within the rebar square and observing the presence of fine sediment (dominant particle < 0.2mm in diameter; sand, silt, or clay) directly below 36 sampling points along two steel bars (i.e., 19 tick marks spaced at even intervals, per bar) welded along the vertical and horizontal axes of the hoop (Kowalski & Richer 2019). Observations were made visually with an underwater viewer (Fieldmaster, Aquaview Underwater Viewer) and by feel. The number of fine-sedimentdominated points were expressed as the percentage of sampling area containing fine sediment (i.e., number of fine sediment observations / 36 total sampling points). Following the methods of Platts et al. (1982), the embeddedness of substrate by fine sediment was also visually estimated from within each steel hoop and expressed on an ordinal scale (1: 0-5% embedded, 2: 5-25 embedded, 3: 25-50% embedded, 4: 50-75% embedded, and 5: 75-100% embedded). After fine sediment and embedded measurements were made, the steel hoop was removed from the rebar square and insect density samples were performed.

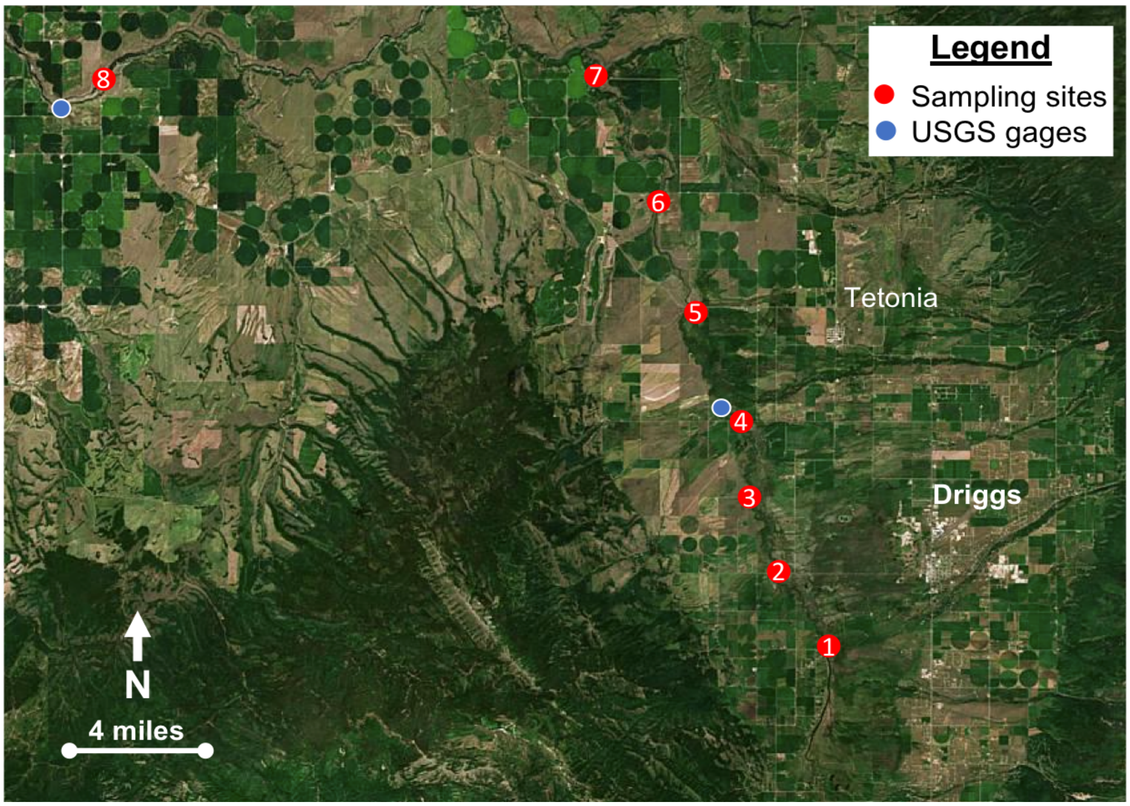

Fig. 1: Map of insect and habitat monitoring sites and USGS gage stations. GPS locations of monitoring sites are as follows: Site 1 – South Bates Bridge: 43.69643, -111.16518; Site 2 – Bates Bridge: 43.7539 -111.2052; Site 3 – Big Eddy: 43.7539,-111.2052; Site 4 – Cache Bridge: 43.78134, -111.20969; Site 5 – Harrop's Bridge 43.82484, -111.23341; Site 6 – River Rim Ranch: 43.86466, -111.25283; Site 7 – Felt Dam: 43.91357; -111.28382; Site 8 – Old Teton Dam 43.91264, -111.53867.

Insects were sampled within each rebar square at the same locations as the sediment measurements. Insects were sampled by spreading a 1 × 1m sampling screen net (1 × 1mm mesh) on the downstream side of the rebar square and disturbing the substrate within the square such that the insects were washed into the net by the current. To do so, all large rocks (> 10 cm in diameter) were removed from the streambed and scrubbed by hand in front of the screen. The smaller substrate was then vigorously disturbed by hand and foot to a depth of ~7cm. Once the substrate was completely disturbed, the contents of the net were brought to shore, washed into polyurethane bins. Insects stuck to the net were picked off with forceps. All insects were then transferred into Whirlpack bags and stored in 100% isopropyl alcohol for preservation. In the lab, insects from each sample were subsampled according to (Barbour et al. 1991), with each subsample representing at least 25% of the total sample and containing at least 100 target species individuals. The number of each target species enumerated per sample, divided by the subsampling percentage (usually 0.25), thus represented its density per $0.25m^2$. All values were then multiplied by four to produce densities of individuals per m^2.

All field efforts were aided by volunteers, who assisted the authors by holding insect sampling nets, processing insect samples, recording data, etc. All volunteers were carefully trained prior to field work, and their actions were closely overseen by the authors in the field.

Temperature and flow measurements

Flow data for each sampling site were measured at 15-minute intervals by US Geological Survey (USGS) gaging stations and were retrieved online (https://waterdata.usgs. gov). Temperature timeseries data were retrieved from temperature loggers (HOBO Pendent) deployed by Friends of the Teton River. Data were used to calculate the following monthly metrics for July, August, and September: mean and mean maximum monthly water temperatures (°C, °F) and mean monthly flows (ft^3/s). For all USGS gaging stations, flow metrics were calculated based on 5 years of historical gaging data from 2019-2023. Long-term temperature data was not available at most sites, however, so only one year of data were used (i.e., 2023).

For sampling site sites close to a gauging or logging station (< 0.25km), temperature and flow metrics of the nearest gage were assumed to accurately represent flow or temperature patterns at the sampling site and were unaltered. For sampling sites that were far from a gauging or logging station (> 0.25 km), temperature and flow metrics were interpolated and extrapolated based on relationships between flow and temperature and elevation. However, the relationship between flow and temperature metrics and elevation were non-linear and sometimes *declined* from upstream to downstream between individual gages. This is likely an effect of local springs, tributary inputs, levels of solar radiation, and groundwater characteristics. To account for local variation, interpolations and extrapolations for each site were based on linear models of elevational differences between the two nearest gaging stations (one upstream and one downstream for interpolation; two upstream from extrapolation), instead of a linear model accounting for the effect of elevation across all sites.

Actual, interpolated, and extrapolated monthly temperature and flow metrics associated with each sampling site were then averaged across the entire summer season (i.e., July-September) to calculate mean, mean maximum monthly summer temperatures, and mean summer flow rates for use in analyses.

Historical mean summer (July-September) flows (ft^3/s) were also calculated for each site and were derived from NHDPlus version 2 (https://nhdplus.com/NHDPlus/NHDPlusV2_home. php), a geospatial, hydrological framework dataset developed by the Environmental Protection Agency and USGS, using ArcGIS (ESRI 2023). Historical mean summer flows were based on data from USGS flow gages from 1971-2000 on unregulated rivers and basin characteristics and precipitation regimes of the Teton River drainage. Differences between the estimated historical flows and contemporary flows (described above) were assumed to be largely due to dewatering from human activities. We, thus, calculated the level of dewatering at each site (hereafter, percent dewatered) by dividing the contemporary mean summer flows by the historical flows and subtracting these values from 1.

Statistical analysis

Analyses for the effects of environmental covariates on densities of each target species were performed in R (R Core Team, 2023) using zero inflated generalized linear mixed effects models (R package: NBZIMM; function: glmm:zimb) (Zhang 2020) due to the high level of dispersion (variance > mean) and excess zeros (0 densities for each species > 20%) of species density data. For each model, site location was included as a random effect because multiple insect density samples were taken at each site.

To ensure that collinearity of predictors had minimal impact on models, we calculated Spearman correlation coefficients between all potential covariates using the R function, cor (R Core Team, 2023). Strong collinearity (Spearman correlation coefficient < -0.33 or > 0.33) was found between nearly all covariates, so we modeled the effect of covariates on each species separately. Covariates with highly non-significant effects (P > 0.10) were dropped from analyses. Pseudo-R2 was calculated for each covariate with a significant (P < 0.05) or near-significant (P > 0.05 < 0.10) effect via the R function, r.squaredGLMM (package: MuMIn) (https://cran.rproject.org/web/pack-ages/MuMIn/index. html), to estimate model fit.

Quality Control and Power Analysis

We performed a series of power analyses to determine the statistical power (i.e., likelihood of finding a statistically significant difference) of our sampling methods to detect a 5%, 10% and 20% annual decline in river-wide densities for each target species over a five-year period with either three, six, or nine samples per site. To do so, we simulated additional data by first calculating theta (i.e., a metric of relative variance used to determine the shape of a negative binomial distribution) and the mean density of each species at each site. For each species, means at each site were then repeatedly multiplied by 0.95, 0.90, and 0.80 (zero to five times), respectively, to simulate a 5%, 10%, and 20% annual decline for each year of the five-year simulated dataset. We then incorporated the theta and reduced mean values for each species-site-year combination into a negative binomial distribution random number generator, with the R function rnbinom (R Core Team, 2023), to generate simulated densities for each species per sampling site and year, with either three, six, or nine samples per site. This resulted in the creation of nine simulated datasets (i.e., three decline levels x three sample size levels). To perform the power analyses, we then repeatedly analyzed the effect of sampling-year on the densities of each species per simulated dataset 100 times using zero inflated negative binomial general linear mixed effects models with site location as a random effect. We then calculated the proportion (i.e., statistical power) of times significant effects were found, given the differences in species declines and samples per site of each simulated dataset.

Results and discussion:

Distribution and density patterns varied among all target species along the Teton River (Table 1). Below, we report and interpret results for each species and set of environmental factors, separately. No comparisons could be made to historical insect assemblages because the few monitoring dataset we were able to locate either used qualitative sampling methods, measured insect densities at different sites, or did not record the relative area that their density samples represent. Analysis of species trends will therefore only be possible after future sampling is performed.

Table 1: Mean densities (individuals/m^2) of each target species found on the Teton River, per site. Least salmonflies, nocturnal stoneflies, and march browns were not found in any samples and their zero densities are not reported in this tale.

Order	Target species	Site 1: South Bates	Site 2: Bates	Site 3: Big Eddy	Site 4: Cache	Site 5: Harrops	Site 6: River Rim Ranch	Site 7: Felt Dam	Site 8: Teton Dam
Stonefly	Giant salmonfly (P. californica)	0	0	0	0	1	34	18	8
Stonefly	Golden stonefly (H. pacifica)	0	0	2	0	4	0	2	0
Stonefly	Skwala. (S. americana)	1	0	0	0	7	1	2	0
Mayfly	Green drake (D. grandis)	23	13	75	26	1	1	6	40
Mayfly	Mahogany dun (Paraleptophlebia sp.)	0	0	0	0	0	0	0	4
Caddisfly	Mother's Day caddis (Brachycentridae)	48	257	221	26	850	32	55	11
Caddisfly	Spotted sedges (Hydropsychidae)	69	41	256	6	13	36	129	47

Giant salmonflies – P. californica

Giant salmonflies were present from Harrops Bridge to Teton Dam (sites 5 to 8), with a mean density of 8.0 individuals per m^2 across the entire stream and highest densities occurring in the canyon at River Rim Ranch and Felt Dam (Table 1). Densities were near significantly correlated with canopy cover ($P = 0.053$), with more salmonflies occurring in reaches with less trees. No correlations were found with any other measured environmental factor, perhaps because temperatures and dewatering levels, which often influence giant salmonfly distributions, were fairly uniform across the study sites and well-within tolerance limits (Birrell & Frakes 2024). However, anecdotal observations suggest that salmonfly distributions on the Teton may be most strongly connected with variation in substrate size (not measured), as sites in the canyon (where densities were highest) appeared to have much larger rocks than in the valley. Indeed, substrate size has been found to be a strong driver of salmonfly densities in other streams, including the Gunnison River, Colorado (Kowalski & Ricker 2019). We assume that this association is also why giant salmonfly densities were *negatively* associated with tree cover (which was also lower in the canyon) – a surprising result given the reliance of salmonflies on leaf material and woody debris for food. Our data provides no evidence that giant salmonflies are of immediate conservation concern on the Teton River, at least in response to the factors we measured.

Least salmonflies – P. badia

Least salmonflies were not found in any samples along the Teton River. Because least salmonflies generally emerge from eggs soon after oviposition and are present in fall samples (e.g., Birrell & Frakes 2024), they should have been found if they are indeed common on this River. We can thus assume that average densities are about 0.0 individuals per m^2 across the entire stream.

Golden stoneflies – H. pacifica
Golden stoneflies were present at sites 3, 4 and 7 (Big Eddy, Harrops, and Felt Dam, respectively), with a mean density of 1 individual per m^2 across the entire Teton River (Table 1). Though densities were quite low at all sites, they were highest at Harrops (site 4). Golden stonefly densities decreased significantly with fine sediment (P = 0.011) and were unaffected by all other habitat factors (Fig. 2).

Results suggest that golden stoneflies are not environmentally stressed on the Teton River, and are likely not of immediate conservation concern. Indeed, short-term heat tolerances of golden stoneflies are quite high for stoneflies and are well-above in-stream values (e.g., Birrell et al. 2021), which should allow them to persist across the stream, even under significant warming. While golden stonefly populations were negatively correlated with fine sediment, fine sediment levels were usually low across the river, and we suspect that they will remain low if levels of dewatering, agriculture, and development remain at current levels. However, if sedimentation increases in the future, golden stonefly densities may begin to decrease. Continual monitoring of fine sediment is thus highly recommended.

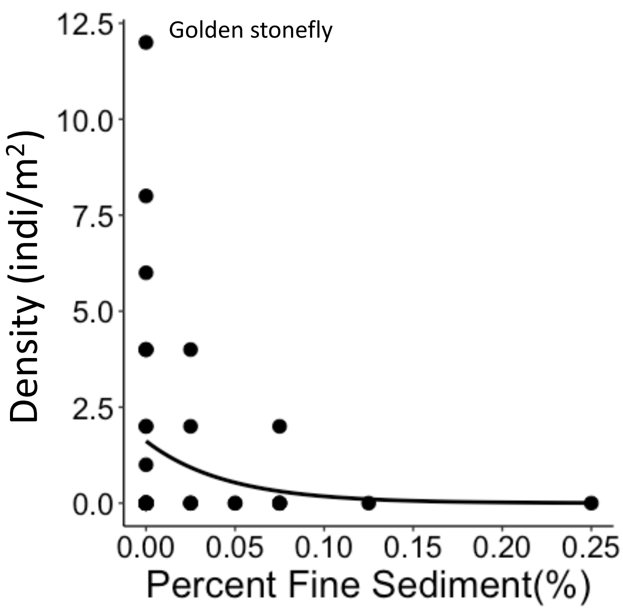

Fig. 2: Scatter plot of the effect of percent fine sediment on the densities of golden stoneflies.

Nocturnal stoneflies – C. sabulosa
Nocturnal stoneflies were not found in any samples from the Teton River. Given the two-year duration of nocturnal stonefly nymphal stages, this species can be encountered in benthic

samples at any time of year, and we can thus assume their population densities are negligible within our sampling area.

Skwala – S. americana
Skwala were present at sites 1 and 5 to 7 (South Bates and Felt to Teton Dam, respectively) (Table 1), and average River-wide densities were 1 individual per m^2, with the highest densities occurring at Harrop's (site 4). Skwala densities were not significantly correlated with any environmental factor we measured, yet were near-significantly correlated with percent fine sediment (P = 0.056) (Fig. 3).

Results suggest that thermal and dewatering conditions are uniformly suitable for Skwala across the Teton River and that this species is not of immediate conservation concern. While skwala densities were negatively correlated with fine sediment, fine sediment was usually low, suggesting that abundant habitat exits within riffles across all sites. Nevertheless, results indicate that additional, future sedimentation may lead to population declines of skwala, especially if percent fine sediment levels rise towards 25%. We advise for continual monitoring of fine sediment across the Teton River.

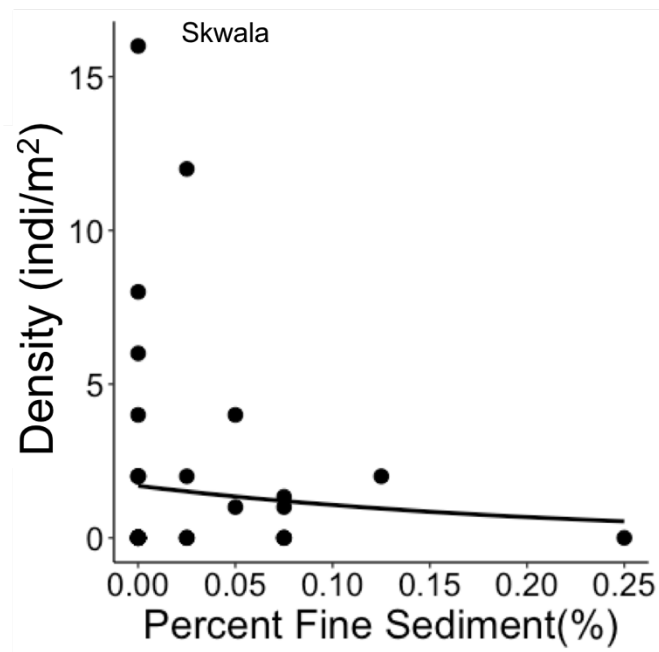

Fig. 3: Scatter plots of the effect of percent fine sediment on densities of skwala.

Green drake – D. grandis
Green drakes were present at all sites, with a mean density of 23 individuals per m^2 across the entire River, with densities spiking sharply at Big Eddy (site 3), at 75 individuals per m^2 on average (Table 1). Green drake densities were not significantly correlated with any environmental factor we measured, suggesting that measured conditions were uniformly suitable across the entire River. Indeed, this is corroborated by similar thermal, dewatering, and

substrate conditions at all sites. Results suggest that green drakes are not of immediate conservation concern.

Mahogany duns – Paraleptophlebia spp.
Mahogany duns were only present at Teton Dam (site 8), having a mean density of < 1 individual per m^2 across the entire River (Table 1). Negligible densities of Mahogany duns across most sites made analysis of environmental predictors challenging, and no significant effects were found.

Our study, may however, underestimate true population sizes because the emergence of this species was underway at the time of collection. In addition, other studies show that mahogany duns prefer slower, siltier habitats (e.g., Relyea et al. 2000), and densities measured in riffles (which tend to be sediment-poor) may further underestimate true population numbers. Because of this, our study may have poor power for detecting true mahogany dun densities and correlations with habitat conditions. Verifying the conservation status of mahogany duns may thus require further monitoring efforts, including collections at a different time of year, outside of the adult emergence.

March browns – *Rhithrogena spp.*
March browns were not found in any samples from the Teton River. Generally, march browns are very small in samples taken in September and it is possible that – if the species is indeed present – most individuals are still in egg form. Future sampling, likely at a different time of year, will thus be required to accurate estimate the densities and distributions of this species on the Teton River.

Mother's Day caddis – Brachycentridae
Mother's Day caddis were present at all sites, and were the most dense of all target species measured. They had an average density of 187 individuals per m^2 across the entire River, with populations peaking at Harrop's Bridge (site 4) at 884 individual per m^2 (Table 1). Mother's Day caddis densities were not significantly correlated with any environmental factor we measured.

Results suggest that environmental conditions we measured are uniformly suitable for Mother's Day caddis across the Teton River and, along with their high densities, that this species is of little conservation concern. Additional monitoring will be necessary to establish trends and probable threats for this dominant family of caddisfly.

Spotted sedges – Hydropsychidae
Spotted sedges were present at all sites and were the second most dense of all target species measured. They had an average density of 75 individuals per m^2 across the entire River. Densities were highest at Big Eddy (site 3), which supported a mean of 256 individuals per m^2 (Table 1). Spotted sedge densities decreased significantly with percent fine sediment ($P = 0.027$), yet are not significantly correlated with other environmental factors we measured (Fig. 4).

As with Mother's Day caddis, high densities of spotted sedges suggest that this species is not of immediate conservation concern. However, negative correlations with fine sediment suggest that this species may begin to decline if additional sedimental becomes severe. A lack of correlation with dewatering and warm temperatures suggests that these conditions are uniformly suitable to Spotted sedges across the River.

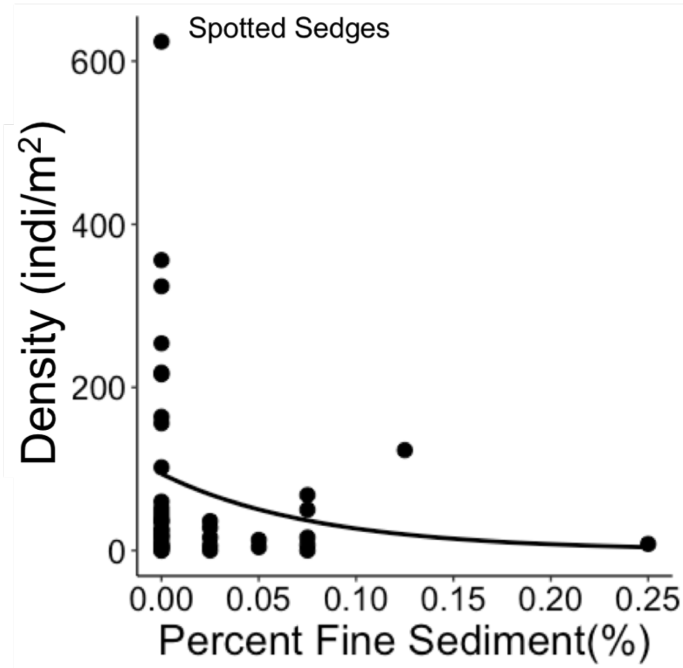

Fig. 4: Scatter plots of the effect of percent fine sediment on densities of spotted sedges.

Environmental factors
Environmental conditions measured on the Teton River in 2023 appear suitable for biodiverse, healthy insect communities and hatches (Table 2).

Across all sites, the Teton River had mean and mean maximum monthly summer temperature of 60.6°F and 68.3°F, respectively. On average, mean and mean maximum monthly summer temperatures were coolest in the middle of the River, likely due to inflows of cold water from tributaries and shading provided by the canyon in the downstream reaches. Nevertheless, the range of mean summer temperature and mean maximum monthly summer temperature among sites was generally low (3.6 °F and 2.7 °F, respectively). Indeed, temperatures appear to be quite cool and are well-below the in-stream thermal tolerance limits of most stoneflies, mayflies, and caddisflies, including target species like giant salmonflies and green drakes (Richards et al. 2013). These results, in combination with infrequent correlations between species densities and temperature in our dataset, indicate that high temperatures are likely not of immediate conservation concern.

Percent dewatering appeared to be at moderately high levels (29%) on the Teton River and suggest significant water loss occur via nearby agricultural and municipal withdrawals. Such dewatering – though lower than on some famous fisheries (e.g., Big Hole River; Birrell et al. 2023) – is concerning because it leads to widespread reductions in habitat for both fish and insects, along with lower levels of stream oxygenation and increased sedimentation. Limiting future dewatering levels will thus be crucial for maintaining both healthy fish and insect communities. Despite obvious potential impacts, however, we were unable to detect statistical effects of dewatering on insect densities in this study because infrequent USGS gaging stations along the River made estimating variation in dewatering among sites impossible. We recommend that future monitoring apply a different approach to estimating dewatering levels.

Fine sediment levels were generally low, with values ranging from 0% to 25% across all samples. Overall, fine sediment made up an average of 3.0% of the substrate sampled, with no fine sediment detected at Teton Dam (site 8) and with the highest amount of fine sediment occurring at South Bates (11%; site 1). These sediment levels are within the tolerance limits of most sediment intolerant insect species, and thus sedimentation is likely not a severe threat to insect populations on the Teton, assuming sediment levels are constant through time (Relyea et al. 2000). However, fine sediment levels were negatively correlated with densities of golden stoneflies, skwala, and Mother's Day caddis, suggesting that sedimentation is likely high enough to limit the distribution and densities of at least some species. We, therefore, urge for careful management practices that limit inputs of additional fine sediment into the Teton River – from agricultural, road maintenance, flow reductions, or human development. Because sediment levels can be highly variable over seasonal and annual scales, a more consistent seasonal sampling of fine sediment may be warranted in the future.

Table 2: Mean values of habitat conditions parameters used in analyses at each site.

Habitat factor	Site 1: South Bates	Site 2: Bates	Site 3: Big Eddy	Site 4: Cache	Site 5: Harrops	Site 6: River Rim Ranch	Site 7: Felt Dam	Site 8: Teton Dam
Mean Summer Temperature (°F)	58.11	60.02	60.59	61.17	61.16	61.15	61.00	61.67
Mean Maximum Monthly Summer Temperature (°F)	66.39	68.66	68.89	69.13	69.07	69.01	68.31	66.97
Mean Summer Flow Rate (cfs)	326.24	338.35	336.38	368.47	450.06	446.83	442.49	659.57
Mean NHDPlus2 Modelled Summer Flow Rate (cfs)	459.49	473.78	473.78	518.97	633.89	629.34	623.22	927.86
Estimated Percent Dewatered (%)	0.29	0.29	0.29	0.29	0.29	0.29	0.29	0.29
Mean Percent Fine Sediment (%)	0.11	0.06	0.00	0.03	0.02	0.02	0.04	0.00

Data Simulations and Power Analysis

Density data were highly variable for all species, with a mean overdispersion and coefficient of variation (CV) (first averaged across each site per species) for all species of 37.0 and 1.04, respectively. Densities of giant salmonflies, spotted sedges, and Mother's Day caddis were the most variable (overdispersion: 10.5, 64.4, and 101.0, respectively; CV: 1.1, 0.958, 0.752, respectively).

Densities of aquatic insects have long-been known to be highly spatially variable, due to the patchy nature of their population structures (Townsend 1989). Natural variability is an integral challenge for insect surveys because it reduces statistical power (i.e., the probability of detecting a statistically significant effect), making it more difficult to detect subtle shifts in insect populations over time. However, data simulations and power analyses show that – despite high variance of insect densities – the sampling effort employed in this study (i.e., six samples per site across nine sites) is capable of detecting moderate to strong shifts in densities of most species (Table 3). Indeed, statistical power was nearly always high (>75%) across all sampling efforts when species declined by 20% per year. More moderate declines of 10% per year also yielded mid-to-high statistical power (50-75%), but only when at least six samples were taken per site. These results suggest that maintaining at least six samples per site will be crucial for detecting moderate to severe insect declines, going forward, and should be employed in future sampling years. However, results also suggest that current methods are likely incapable of detecting subtle changes in insect assemblages, as 5% annual declines never yielded high statistical power, even at nine samples-per-site. This highlights the need for updating project methods to better allow for the detection of subtle trends in insect densities.

Because data variability can also be exacerbated by sampling techniques that introduce bias or randomness (Barbour et al. 1999), we recommend that future sampling efforts employ methods that reduce sources of variation and thereby increase statistical power. In future years, using a closed-system collection net, such as a Hess sampler, instead of the open face sieve employed in this study, may reduce variation by preventing insects from being washed into the net from outside the sampling area or out of the net during streambed disturbance. Doing so will likely allow statistical power to increase, such that subtle to moderate declines will be more readily detected (Kraemer & Blasey 2015). Power analyses should we performed again after the second year of data collection with the new, closed system collection net to reevaluate the number of sites and samples-per-site necessary for detecting shifts in insect density.

Conclusion:

While continuing this monitoring program into the future will be key to confidently establish trends of target species and drivers of potential declines, our study has important conservation implications after only one year of data collection. In general, habitat conditions, including mean and maximum summer temperatures and levels of dewatering, appear to be low and within healthy levels for supporting biodiverse, cold-water aquatic insect communities. Frequent positive correlations between target species densities and fine sediment, however, suggest that sedimentation may be the most likely, immediate threat to aquatic insects on the Teton River. However, our data also shows that fine sediment levels are nearly ubiquitously low, and large changes in land use and heavy dewatering would likely be necessary to increase fine sediment to harmful levels. Continued data collection will be key to validating predictions, establishing trends in target insect populations and environmental factors over time, and improving conservation recommendations. Because our data are broadly indicative of a healthy aquatic insect community and high water quality, we suggest future monitoring efforts should

Species	% Annual Decline	Samples per site	Relative Effort	Power: 95% CI	Power: 90% CI
Giant salmonfly (P. californica)	5	3	1.00	0.24	0.32
		6	0.56	0.27	0.43
		9	0.53	0.38	0.46
	10	3	1.88	0.45	0.54
		6	1.38	0.66	0.72
		9	1.01	0.73	0.85
	20	3	3.58	0.86	0.92
		6	2.00	0.96	0.98
		9	1.39	1.00	1.00
Golden stonefly (H. pacifica)	5	3	0.88	0.21	0.28
		6	0.58	0.28	0.33
		9	0.51	0.37	0.47
	10	3	1.54	0.37	0.43
		6	0.88	0.42	0.51
		9	0.72	0.52	0.61
	20	3	1.35	0.65	0.74
		6	1.11	0.80	0.88
		9	1.98	0.95	0.96
Skwala (S. americana)	5	3	1.13	0.27	0.40
		6	0.63	0.30	0.39
		9	0.56	0.40	0.49
	10	3	1.71	0.41	0.48
		6	1.48	0.71	0.74
		9	1.01	0.73	0.80
	20	3	1.67	0.80	0.83
		6	1.31	0.94	0.97
		9	2.08	1.00	1.00
Green drake (D. grandis)	5	3	0.63	0.15	0.25
		6	0.77	0.37	0.46
		9	0.61	0.44	0.59
	10	3	2.38	0.57	0.73
		6	1.94	0.93	0.95
		9	1.38	0.99	0.99
	20	3	2.06	0.99	1.00
		6	1.39	1.00	1.00
		9	2.08	1.00	1.00
Mahogany dun (Paralepophlebia)	5	3	1.54	0.37	0.44
		6	0.92	0.44	0.54
		9	0.71	0.51	0.54
	10	3	2.08	0.50	0.64
		6	1.27	0.61	0.69
		9	0.93	0.67	0.79
	20	3	1.52	0.73	0.79
		6	1.29	0.93	0.94
		9	2.04	0.98	0.98
Mother's Day caddis (Brachycentridae)	5	3	0.71	0.17	0.24
		6	0.42	0.20	0.34
		9	0.50	0.36	0.44
	10	3	2.17	0.52	0.67
		6	1.52	0.73	0.82
		9	1.15	0.83	0.90
	20	3	2.04	0.98	1.00
		6	1.39	1.00	1.00
		9	2.08	1.00	1.00
Spotted sedge (Hydropsychidae)	5	3	0.63	0.15	0.23
		6	0.44	0.21	0.36
		9	0.43	0.31	0.44
	10	3	1.54	0.37	0.47
		6	1.56	0.75	0.84
		9	1.17	0.84	0.90
	20	3	0.44	0.93	0.97
		6	0.43	1.00	1.00
		9	1.94	1.00	1.00

Table 3: Results of power analyses, showing the effect of annual decline and samples per site on statistical power (probability of detecting a significant effect) and relative effort (statistical power/total sites) using both 95% and 90% confidence intervals. Colors represent different levels of statistical power – red: very low (< 25%), low (> 25% < 50%), moderate (> 50% < 75%), and high (> 75%). Power analyses were based on a simulated 5%, 10%, and 20% annual decline for each species over five years, with means and standard deviations based on the 2023 dataset, reported here.

also investigate patterns in nearby tributaries with more obvious human pressure. We also suggest incorporating additional environmental factors into the analysis like water chemistry and nutrients pollution, and/or expanding sampling sites upstream to capture a wider suite of environmental conditions that may be influencing target species densities. In addition, the absence of several target species (e.g., nocturnal stoneflies, march browns, etc.) suggests that incorporating additional target species – such as blue winged olives (Baetis spp.) and tricos (Tricorythodes spp.) – should be included as replacements in future sampling years. Lastly, while power analyses showed that current methods will provide sufficient statistical power for detecting moderate to strong insect declines, they also suggest that they are likely insufficient for detecting more subtle trends that may occur in the near future. We therefore recommend that future sampling prioritize sampling techniques that reduce sampling error and variance (e.g., by using Hess samplers), which should likely enhance the ability of the monitoring program to detect subtle changes in insect densities, going forward.

Acknowledgements and data availability:
We thank the Teton Valley Chapter of Trout Unlimited for funding this monitoring program. We appreciate Maggie Heumann for consulting on project design and Friends of the Teton River for providing sampling data for 2022-2023. We also thank all volunteers to who helped us sample insects and measure habitat conditions in the field. All data collected and used in this work will be made available upon request. For data inquires, contact The Salmonfly Project at conservation@salmonflyproject.org.

References:
Albertson, L. K., Briggs, M. A., Maguire, Z., Swart, S., Cross, W. F., Twining, C. W., … Walters, D. M. (2022). Dietary composition and fatty acid content of giant salmonflies (Pteronarcys californica) in two Rocky Mountain rivers . *Ecosphere*, *13*(1), 1–10.

Allan, D. J., & Castillo, M. M. (2007). *Stream ecology: structure and function of running waters*. Dordrecht: Springer Science & Business Media.

Anderson, H. E., Albertson, L. K., & Walters, D. M. (2019). Water temperature drives variability in salmonfly abundance, emergence timing, and body size. *River Research and Applications*, *35*(7), 1013–1022.

Armour, C., Duff, D., & Elmore, W. (1994). The effects of livestock grazing on western riparian and stream ecosystem. *Fisheries*, *19*(9), 9-12.

Barbour, M. T., Faulkner, C., & Gerritsen, J. (1991). Rapid Bioassessment Protocols For Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish Second Edition http://www.epa.gov/OWOW/monitoring/techmon.html By: Project Officer.

George Gibson (Ed). Biological Criteria: Research and Regulation, Proceedings of a Symposium (December 1213, 1990: Arlington, Virginia), 174.

Birrell, J. H., & Woods, H. A. (2023). Going with the flow – how a stream insect, Pteronarcys californica, exploits local flows to increase oxygen availability. *Journal of Experimental Biology*.

Birrell, J. H., Frakes, J. I., Shah, A. A., & Woods, H. A. (2023). Mechanisms underlying thermal breadth differ by species in insects from adjacent but thermally distinct streams – A test of the climate variability hypothesis. *Journal of Thermal Biology*, *112*(December 2022), 103435.

Birrell, J. H., Meek, J. B., & Nelson, C. R. (2019). Decline of the Giant Salmonfly Pteronarcys Californica Newport , 1848 (Plecoptera : Pteronarcyidae) in the Provo River, Utah, USA. *Illiesia*, *15*(5), 83–97.
Birrell, J. H., Meek, J. B., & Nelson, C. R. (2019). Decline of the Giant Salmonfly Pteronarcys Californica Newport , 1848 (Plecoptera : Pteronarcyidae) in the Provo River, Utah, USA. *Illiesia*, *15*(5), 83–97.
Birrell, J. H., Shah, A. A., Hotaling, S., Giersch, J. J., Williamson, C. E., Jacobsen, D., & Woods, H. A. (2020). Insects in high-elevation streams: Life in extreme environments imperiled by climate change. *Global Change Biology*, *26*(12), 6667–6684.

Ball, B. (2023). Bureau of Reclamation to reduce water flow from Jackson Lake Dam starting May 10. Wyoming Game and Fish Department. Retrieved from: https://wgfd.wyo.gov/ News /Bureau-of-Reclamation-to-reduce-water-flow-from-Ja

Bonavist, T. (2023). Where have all the mayflies gone?. *River Reporter*. Accessed from: https://riverreporter.com/stories/where-have-all-the-mayflies-gone,80920

Bunte, K., & Abt, S. R. (2001). *Journal of the American Water Resources Association Sampling Frame for Improving Pebble Count Accuracy in Coarse Gravel-Bed Streams*. *37*(4).

DeWalt, R. E., Favret, C., & Webb, D. W. (2005). Just How Imperiled Are Aquatic Insects? A Case Study of Stoneflies (Plecoptera) in Illinois. *Annals of the Entomological Society of America*, *98*(6), 941–950.

Elder, J. A., & Gaufin, A. R. (1973). Notes on the occurrence and distribution of pteronarcys california newport (plecoptera) within streams. *Great Basin Naturalist*, *33*(4), 218–220.

Environmental Systems Research Institute (ESRI). (2023). *ArcGIS Release 10.8*. Redlands, CA.

Fox, J. W. (2013). The intermediate disturbance hypothesis should be abandoned. *Trends in Ecology and Evolution*, *28*(2), 86–92.

Frakes, J. I., Birrell, J. H., Shah, A. A., & Woods, H. A. (2021). Flow increases tolerance of heat and hypoxia of an aquatic insect. *Biology Letters*, *17*(5).

Giersch, J. J., Hotaling, S., Kovach, R. P., Jones, L. A., & Muhlfeld, C. C. (2017). Climate-induced glacier and snow loss imperils alpine stream insects. *Global Change Biology*, *23*(7), 2577–2589.

Huff, D. D., Hubler, S. L., Pan, Y. D., & Drake, D. L. (2006). Detecting shifts in macroinvertebrate assemblage requirements: Implicating causes of impairment in streams. *Oregon Department of Environmental Quality*, Portland, OR, 38.

Isaak, D. J., Wollrab, S., Horan, D. L., & Chandler, G. (2012). Climate change effects on stream and river temperatures across the northwest U.S. from 1980–2009 and implications for salmonid fishes. *Climatic Change*, *113*(2), 499–524.

Kiewiet, M. (2023). Trout numbers at historic lows on Big Hole River. *Montana Standard*. Accessed from: https://mtstandard.com/outdoors/data-shows-historical-trout-decline-in-big-hole-river/article_f0b292e2-f8c2-11ed-9c24-13673c2b81c7.html

Kowalski, D. A., & Richer, E. E. (2020). Quantifying the habitat preferences of the stonefly Pteronarcys californica in Colorado. *River Research and Applications*, *36*(10), 2043–2050.

Lento, J., Morin, A. (2014). Filling the gaps in stream size spectra: using electroshocking to collect large macroinvertebrates. *Hydrobiologia*, *732*, 1-17.

Macadam, C. R., & Stockan, J. A. (2015). More than just fish food: Ecosystem services provided by freshwater insects. *Ecological Entomology*, *40*(S1), 113–123.

Merritt, R. W., Cummins, K. W., & Berg, M. B. (2008). *An introduction to the aquatic insects of North America. Kendall Hunt, 1996.* Kendall.

Montana Fish Wildlife & Parks. (2015). Montana's State Wildlife Action Plan*. Montana Fish Wildlife & Parks*, Helena, MT.

Nehring, R. B., Heinold, B., & Pomeranz, J. (2011). Colorado River Aquatic Resources Investigations Federal Aid Project F-237R-18. *Colorado Division of Wildlife*.

Platts, W. S., Megahan, W. F., & Minshall, G. W. (1982). Methods for evaluating stream, riparian, and biotic conditions (Vol. 138). *US Department of Agriculture, Forest Service.* Intermountain Forest and Range Experiment Station, Ogden, UT.

R Core Team. 2023 R: a language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing.

Relyea, C. D., Minshall, G. W., & Danehy, R. J. (2000). Stream insects as bioindicators of fine sediment. *Watershed Management Conference*.

Richards, D. C., Bilger, M., & Lester, G. (2013). Development of Idaho Macroinvertebrate Temperature Occurrence Models. *EcoAnalysts, Inc.*, 1–71.

Sautner, S. (2023). Dude, where's my hatch?. *Patagonia*. Accessed from: https://www.patagonia.com/stories/dude-wheres-my-hatch/story-136295.html

Sheldon, A. L. (1980). Resource division by perlid stoneflies (Plecoptera) in a lake outlet ecosystem. *Hydrobiologia*, *71*, 155-161.

Stagliano, D. (2013). Evaluation of salmonflies in Montana's rivers : are statewide populations really declining? *Montana Natural Heritage Program*, 1–29.

Stagliano, D. M. (2021). *Lower Big Hole River 2021 Macroinvertebrate Density & Community Study*.

Stagliano, D. M., & Petersen, A. (2016). Macroinvertebrate communities in the Big Hole River (Montana): a comparison of sites sampled ~ 50 years apart. *Intermountain Journal of Sciences*, *22*(1), 1–3.

Stepanian, P. M., Entrekin, S. A., Wainwright, C. E., Mirkovic, D., Tank, J. L., & Kelly, J. F. (2020). Declines in an abundant aquatic insect, the burrowing mayfly, across major North American waterways. *Proceedings of the National Academy of Sciences of the United States of America*, *117*(6), 2987–2992.

USDA Forest Service, National Headquarters. (2011). Biological Assessment of the United States Department of Agriculture National Forest System Land Management Planning Rule.

van Vliet, M. T., Franssen, W. H., Yearsley, J. R., Ludwig, F., Haddeland, I., Lettenmaier, D. P., & Kabat, P. (2013). Global river discharge and water temperature under climate change. *Global Environmental Change*, *23*(2), 450-464.