Monitoring Aquatic Insect Densities and Environmental Factors on the Upper Snake River, Wyoming – 2023 Report

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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 upper Snake River in north western Wyoming. Historically, few systematic efforts have been made to monitor aquatic insect populations on the upper snake River. The primary goals of our project are to annually track: i) the population densities of target insect species that represent the most abundant and important insects on the River and ii) 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 on-going aquatic insect declines and shows that temperature, flow, and sediment conditions on the upper Snake appear to be within healthy levels for supporting an abundant, biodiverse insect community. Continual monitoring will be necessary, however, for tracking any future shifts in insect assemblages and environmental conditions and for identifying conservation priorities to protect the health and biodiversity of this renowned fishery.

Data collected September, 2023; Report submitted March, 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 upper Snake River, Wyoming. The upper Snake River is well-known for its wild scenery and abundant native cutthroat trout, which draw tourists and anglers from around the world. However, anglers also report changes in the strength and timing of insect hatches due to altered flow regimes from the Jackson Lake dam and warmer water temperatures. Indeed, anecdotal records show that some insect species have disappeared altogether, including the giant salmonfly, which used to be abundant in the upper reaches of the River, near Moose, but has since disappeared (Bruun 2024). Unfortunately, historic scientific data on insects of the upper Snake are sparse, and there are currently no on-going, 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 upper Snake River, highlighting the need for additional monitoring.

To help fill this gap, we initiated an aquatic insect and habitat monitoring program across eight sites on the upper Snake River, from Jackson Lake to Palisades Reservoir. 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 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 upper Snake River.

Methods:

Target species:

We selected eleven target taxa of aquatic insects to monitor on the upper Snake 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), with the exception of giant salmonflies, are among the most dominant insect hatches on the upper Snake and are thus of primary importance for both the ecology of the River and its recreational, cultural, and economic value for anglers. Giant salmonflies were included because anecdotal evidence suggests they may have been previously abundant and have since declined (Bruun 2024). 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 upper Snake, 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.

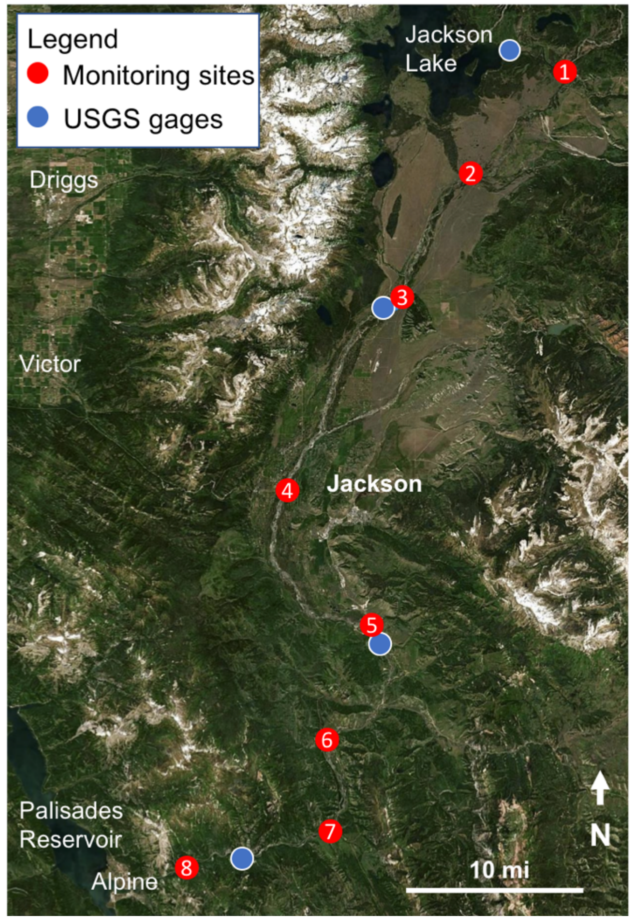

Fig. 1: Map of insect and habitat monitoring sites and USGS gage stations. GPS locations of monitoring sites are as follows: Site 1 – Moran: 43.846173, -110.518714; Site 2 – Deadman's Bar: 43.759590, -110.628010; Site 3 – Moose: 43.65520, -110.71373; Site 4 – Wilson Bridge: 43.49996, -110.83980; Site 5 – South Park: 43.38534, -110.74521; Site 6 – Pritchard Boat Ramp: 43.29154, -110.79003; Site 7 – Elbow Boat Ramp: 43.21284, -110.79036; Site 8 – Sheep Gulch Boat Ramp: 43.19529, -110.92522.

Monitoring sites:

We collected insect target species and measured fine sediment and embeddedness levels at seven sites along the length of the upper Snake between Jackson Lake and Palisades Reservoir, Wyoming from September 12-26, 2023 (Fig. 1). Sites were riffled reaches with cobble-dominated substrate. 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 haphazardly-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-sediment-dominated 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.

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
Temperature and 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). 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 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 two years of data were used from 2022-2023, except for the gage closest to site 8 (Alpine), which had only one year of data available (2023).

For sampling site 3 (Moose), which was close to a gaging station (< 0.25km), temperature and flow metrics of the nearest gage were assumed to accurately represent flow and temperature patterns at the sampling site and were unaltered. For sampling sites that were far from a gaging 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 and 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 upper Snake 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. Higher percentages, thus, represent higher levels of dewatering and these were used in analyses.

Statistical analysis
Analyses for the effects of environmental covariates on densities of each target species were performed in R (R Core Team, 2023) using generalized linear mixed effects models (R package: lme4; function: glmer) (https://cran.r-project.org/web/packages/lme4/index.html). For each model, site location was included as a random effect because multiple insect density samples

were taken at each site. Due to the right-skewed nature of the insect density data, which is common for count data, models were customized to best fit a Poisson distribution.

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) and excluded strongly collinear variables (Spearman correlation coefficient < -0.33 or > 0.33) from the same models. Strong collinearity was found between percent dewatering and mean maximum monthly summer temperature, and these were never included in the same models.

To perform the analyses, the effect of each environmental covariate on each species' densities was first modeled in isolation. Covariates with non-significant effects were dropped from subsequent analyses on that species. Estimated embeddedness was dropped from all analyses because it never had a significant effect. In subsequent models, remaining covariates (i.e., those with significant effects) were modeled additively together. When strongly colinear predictors were both found to be significant, they were included in separate models. Pseudo-R^2 and Akaike information criterion (AIC) were then calculated via the R functions, *r.squaredGLMM* (package: MuMIn) (https://cran.r-project.org/web/packages/MuMIn/index.html) and *aictab* (package: AICcmodavg) (https://cran.r-project.org/web/packages/AICcmodavg/index.html), respectively.

Results and discussion:

Distribution and density patterns varied among all target species along the upper Snake 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 per site.

Order	Target species	Site 1: Moran	Site 2: Deadman's	Site 3: Moose	Site 4: Wilson Bridge	Site 5: South Park	Site 6: Pritchard	Site 7: Elbow	Site 8: Sheep Gulch
Stonefly	Giant salmonfly (P. californica)	0	0	5	0	0	2	0	0
Stonefly	Least salmonfly (P. badia)	64	11	53	5	26	0	3	0
Stonefly	Golden stonefly (H. pacifica)	2	3	27	13	16	23	0	34
Stonefly	Nocturnal stonefly (C. sabulosa)	2	3	32	8	30	6	8	0
Stonefly	Skwala (S. americana)	28	29	27	19	19	25	34	40
Mayfly	Green drake (D. grandis)	10	33	32	37	217	40	39	16
Mayfly	Mahogany dun (Paraleptophlebia sp.)	1	0	0	8	0	1	1	4
Mayfly	March brown (Rhithrogena sp.)	809	56	288	27	92	11	95	46
Caddisfly	Mother's Day caddis (Brachycentridae)	6	4699	2357	3757	2002	109	225	30
Caddisfly	Spotted sedges (Hydropsychidae)	66	295	1339	539	1175	622	2587	926

Giant salmonflies – P. californica

Giant salmonflies were present at only Moose and Pritchard (sites 3 and 6), with a mean density of 0.9 individuals per m^2 across the entire stream. Because of their infrequent presence, it was not possible to analyze correlations between giant salmonfly densities and habitat factors.

Why giant salmonflies are not more abundant on the upper Snake is not obvious, as anecdotal records suggest they were historically abundant near Moose and in the lower canyon. Habitat conditions appear to be conducive, with large substrates, cool water temperatures, low levels of dewatering, and low sedimentation (Table 2) and with dense populations existing in several nearby tributaries. Indeed, giant salmonflies often co-occur with least salmonflies, which were abundant in our samples and which feed on similar detrital food resources. We speculate that giant salmonfly populations may be suppressed on the upper Snake due to intense floods that regularly carve new channels and displace mature cobble bars and substrates that salmonflies need to survive over their long, four year lifespan. Frequent flooding may have less of an effect on short-lived species – all other target species require one or two years to mature – as these may be able to reproduce over multiple generations and grow in population size in between disturbance events (e.g., Fox 2013).

Least salmonflies – P. badia

Least salmonfly were found at Moran to South Park (sites 1-5) and at Elbow Boat Ramp (Site 7), with an average densities of 20.3 individual per m^2 across the entire river (Table 1). Relatively high densities of least salmonflies are interesting because, as stated above, least salmonflies often occur with giant salmonflies – which were rare in our samples. If giant salmonflies have declined since historic levels, as anecdotal reports suggest, it's possible that least salmonflies have become more abundant due to less resource competition, as was reported to occur on the Provo River, UT (Gaufin 1973).

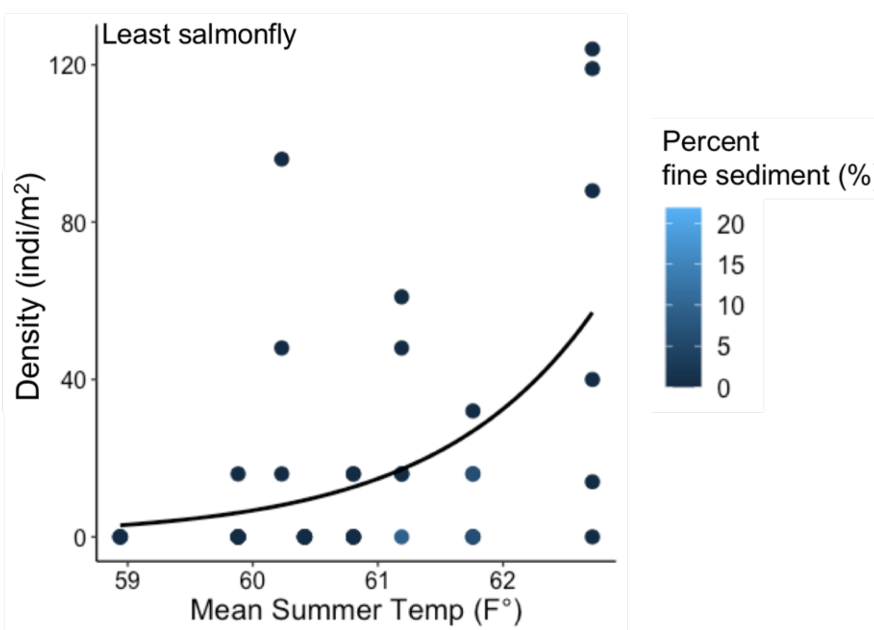

Fig. 2: Scatter plot of the effect of mean summer temperature and percent fine sediment on the densities of least salmonflies.

Least salmonfly densities were significantly affected by both mean summer temperature and percent fine sediment (P < 0.001), with more individuals in warmer reaches and those with less sediment (Fig. 2) (Table S1). In total, results suggest that least salmonflies are likely not of immediate conservation concern. While densities were found to be lower in samples with more fine sediment, fine sediment levels on the upper Snake were nearly ubiquitously low and will likely remain low as long as future flows remain high (Table 2). However, if flows decline, causing sedimentation to increase, least salmonfly densities may decrease in the future. Such changes could be offset, however, by accompanying increases in water temperatures (Isaak et al. 2012), which may cause least salmonflies to become *more* abundant.

Golden stoneflies – H. pacifica

Golden stoneflies were present at all sites, besides Elbow Boat Ramp (site 7), with a mean density of 14.75 individuals per m^2 (Table 1). Golden stonefly densities decreased significantly with fine sediment (P < 0.001) and were unaffected by all other habitat factors (Fig. 3) (Table S2).

Results suggest that golden stoneflies are thermal generalists, as has been found in other systems (Sheldon 1980; Birrell et al. 2022), and are likely not of immediate conservation concern. Indeed, short-term heat tolerances of golden stoneflies are quite high for stoneflies and may allow them to persist, even if temperatures rise significantly in the future (e.g., Birrell et al. 2021). While golden stonefly populations were negatively correlated with fine sediment, fine sediment levels were nearly ubiquitously low across the river (Table 2), and we suspect that they will remain low if levels of dewatering remain at current levels. However, if flows decline and sedimentation increases in the future, golden stonefly densities may begin to decrease.

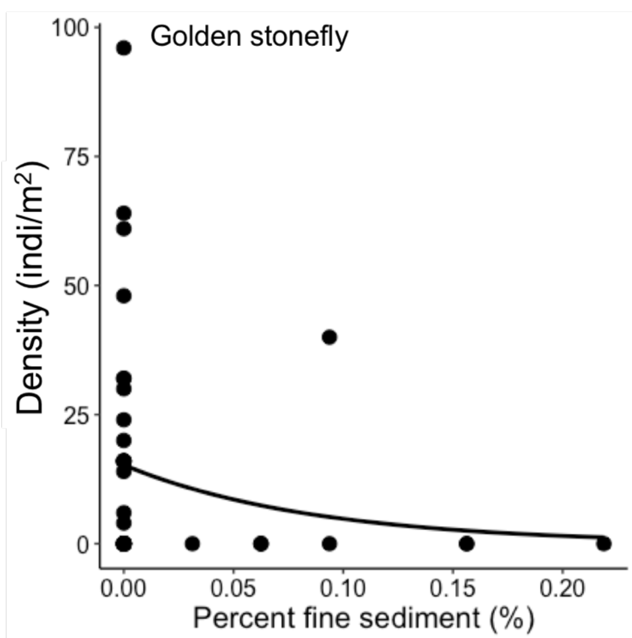

Fig. 3: Scatter plot of the effect of percent fine sediment on the densities of golden stoneflies.

Nocturnal stoneflies – C. sabulosa

Nocturnal stoneflies were present at all sites, besides Sheep Gulch (site 8), with a mean density of 11.13 individuals per m^2 (Table 1). Nocturnal stonefly densities were significantly affected by mean maximum summer water temperatures ($P < 0.001$), with more nymphs occurring at sites with higher maximum summer (Fig. 4) (Table S3).

The positive effect of maximum summer temperature suggests that nocturnal stoneflies are warm-water specialists and that any historical warming has not limited current populations. These results, along with the null effect of other environmental factors, suggest that nocturnal stoneflies are not of immediate conservation concern.

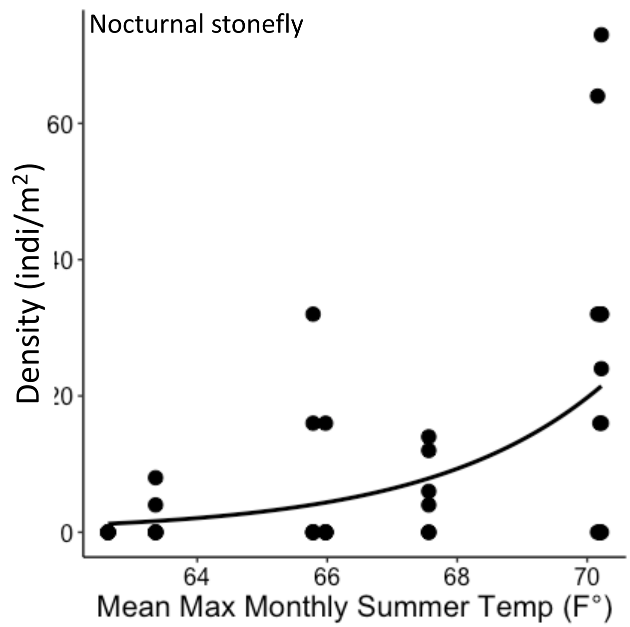

Fig. 4: Scatter plot of the effect of mean maximum monthly summer temperature on the densities of nocturnal stoneflies.

Skwala – S. americana

Skwala were present at all sites, with a mean density of 27.6 individuals per m^2. Densities were highest in the lower reaches of the River, with the highest densities occurring at Sheep Gulch (site 8), at an average of 40 individuals per m^2 (Table 1). Skwala densities significantly decreased with mean maximum monthly summer temperature (model 1: $P < 0.001$) (Fig. 5a) and significantly increased with percent dewatering (model 2: $P = 0.006$) (Fig. 5b) (Table S4).

Positive relationships between skwala densities and dewatering likely reflect skwala's high tolerance and perhaps even preference for slower-flowing habitats (Birrell et al. 2022). Indeed, skwala are known to be one of the most prolific hatches on the lower reaches of the Bitterroot River, Montana – also monitored by SFP – which is much sandier and slower-flowing than the upper Snake. Our results suggest that if dewatering from human development and agriculture become more prevalent on the upper Snake, populations of this species could respond positively, at least until disturbances become severe. Results also suggest, however, that future warming – a common corollary of dewatering – could cause this species to become less abundant. Taken together, these results suggest that skwala are not of immediate conservation concern.

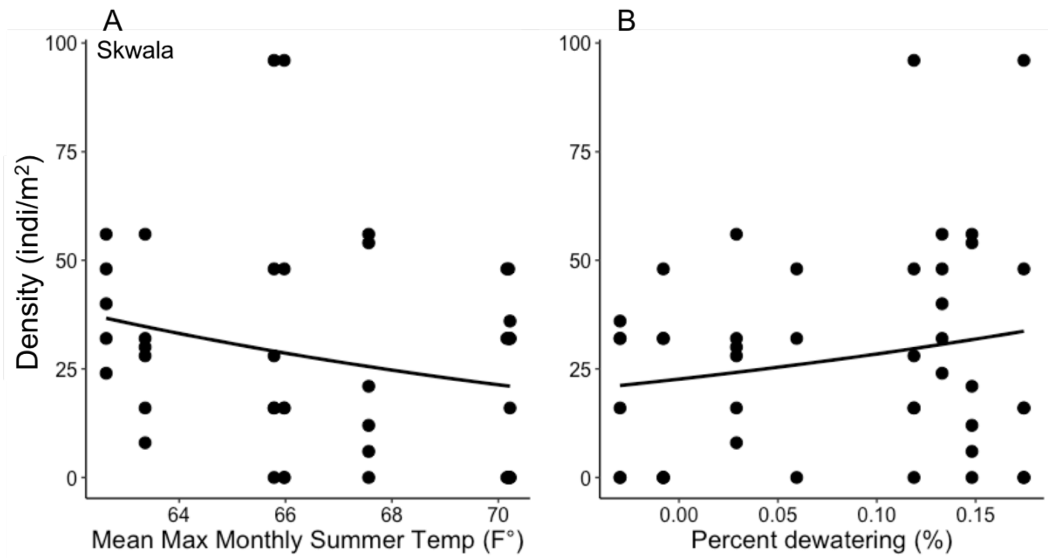

Fig. 5: Scatter plots of the effect of mean maximum monthly summer temperature (A) and percent dewatering (B) on the densities of skwala.

Green drake – D. grandis

Green drakes were present at all sites, with a mean density of 53.0 individuals per m^2 across the entire River, with densities spiking sharply at South Park (site 5), at 217 individuals per m^2 (Table 1). Green drake densities significantly increased with both mean maximum monthly summer temperature and percent fine sediment ($P < 0.001$) (Fig. 6) (Table S5).

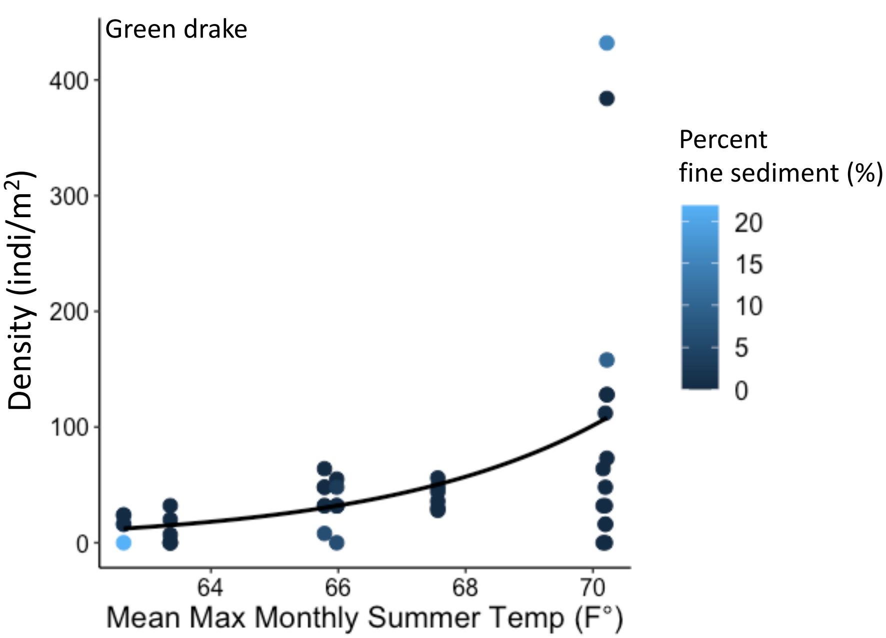

Fig. 6: Scatter plot of the effect of mean maximum monthly summer temperature and percent fine sediment on the densities of green drakes.

Results suggest that green drakes are of little conservation concern on the upper Snake. Indeed, results suggest that current populations may be limited by *cool* water temperatures, and some degree of warming may lead to *higher* densities of this species, at least until warming becomes severe. Likewise, green drakes may become more abundant with slight to moderate increases in sand or silt – previous research shows that green drakes are only moderately sensitive to fine sediment and appear to tolerate substrates covered by 30-50% fine sediment, much higher than any reach surveyed in this study (Relyea et al. 2000).

Mahogany duns – Paraleptophlebia spp.

Mahogany duns were present at Moran (site 1), Wilson Bridge (site 4), and from Pritchard to Sheep Gulch (sites 6-8) and had a mean density of 1.9 individuals per m^2 across the entire River (Table 1). Mahogany dun densities significantly increased with percent fine sediment ($P < 0.001$) (Fig. 7) (Table S6). No significant correlations were found with other predictors.

Results suggest that mahogany duns are of little conservation concern on the upper Snake and that populations may increase with further sedimentation in riffles areas (see also Birrell et al. 2022). Our study, may however, underestimate true population sizes because the emergence of this species was underway at the time of collection. In addition, and in agreement with our results, 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 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.

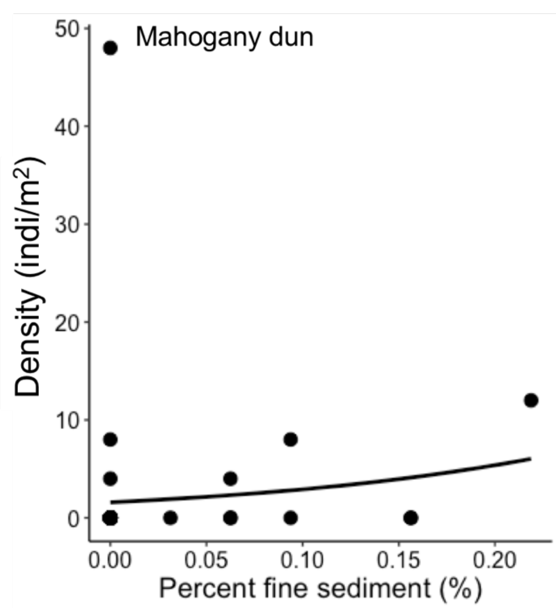

Fig. 7: Scatter plot of the effect of percent fine sediment on the densities of mahogany duns.

March browns – *Rhithrogena spp.*

March browns were present at all sites, with a mean density of 178.0 individuals per square meter across the River and a notable spike in densities at Moran (site 1), with 809 individuals per square meter (Table 1). Densities of march browns were significantly and negatively affected by sedimentation ($P < 0.001$) (Fig. 8) (Table S7). No significant correlations were found with other predictors.

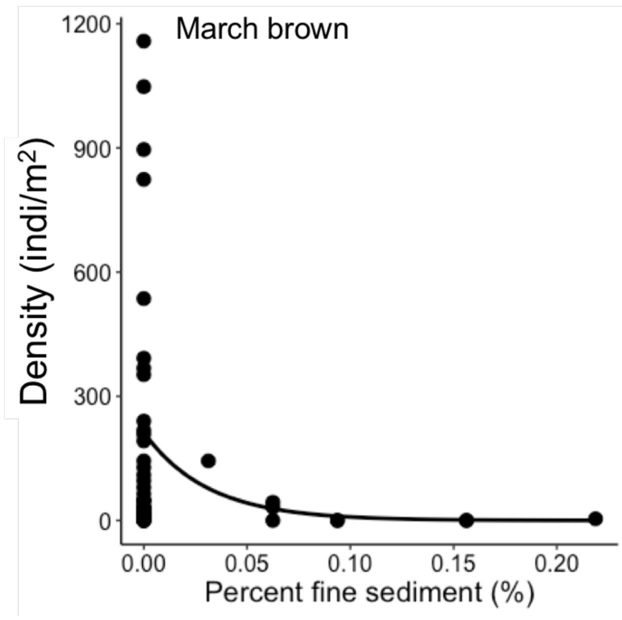

Fig. 8: Scatter plot of the effect of percent fine sediment on the densities of march browns.

Results suggest that march brown are thermal generalists, at least within the temperature ranges present on the upper Snake, and are likely not of immediate conservation concern. While march browns populations were strongly negatively correlated with fine sediment, fine sediment levels were nearly ubiquitously low across the river (Table 2), and we suspect that they will remain low if levels of dewatering remain at current levels. However, if flows decline and sedimentation increases in the future, march browns may begin to decline.

Mother's day caddis – Brachycentridae

Mother's day caddis were present at all sites, though few individuals were found at Moran (site 1). This species was the most dense of all target species measured and had a mean density of 1648.2 individuals per m^2 across the entire River. Densities were highest in mid to upper reaches, with the highest densities occurring at Deadman's Bar (site 2), which supported a mean of 4699 individuals per m^2 (Table 1). Mother's day caddis densities increased significantly ($P < 0.001$) with mean maximum monthly summer temperatures and decreases with fine sediment (Fig. 9) (Table S8).

Due to high densities of mother's day caddis, it's clear that this species is not of immediate conservation concern. Indeed, given the positive correlation between this species density and maximum water temperature, a degree of warming may cause this species to become even more prevalent. However, significant negative correlations with fine sediment suggest that additional sedimentation on the upper Snake may cause populations to decrease. We suspect this is unlikely, however, if levels of dewatering remain at low levels going forward.

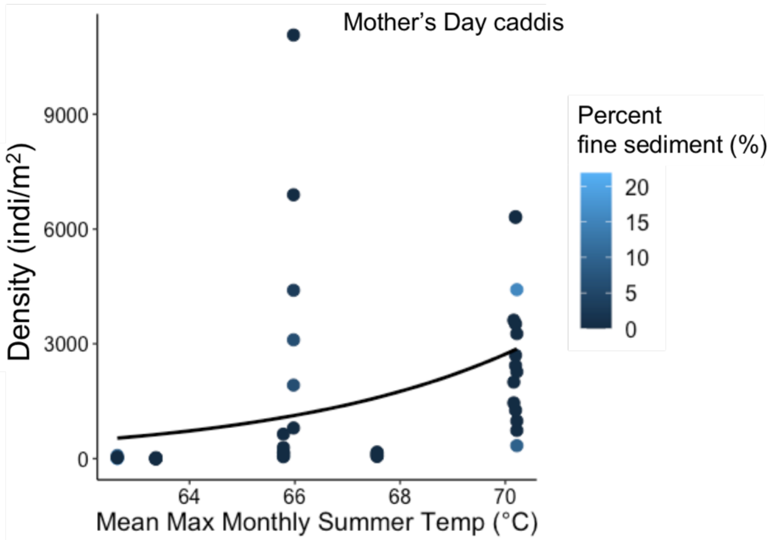

Fig. 9: Scatter plot of the effect of mean maximum monthly temperature and percent fine sediment on densities of mother's day caddis.

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 943.6 individuals per m^2 across the entire River. Densities were highest in mid to lower reaches, with the highest densities occurring at Elbow Boat Ramp (site 7), which supported a mean of 2587 individuals per m^2 (Table 1). Spotted sedge densities decreased significantly with mean summer temperatures and percent fine sediment (Fig. 10) (Table S9).

As with mother's day caddis, high densities of spotted sedges suggest that this species is not of immediate conservation concern. However, correlations with mean summer temperature and fine sediment suggest that any future warming and sedimentation – resulting from climate change, land use change, and dewatering – may cause populations to decline. We recommend continued monitoring of spotted sedges to determine future trends.

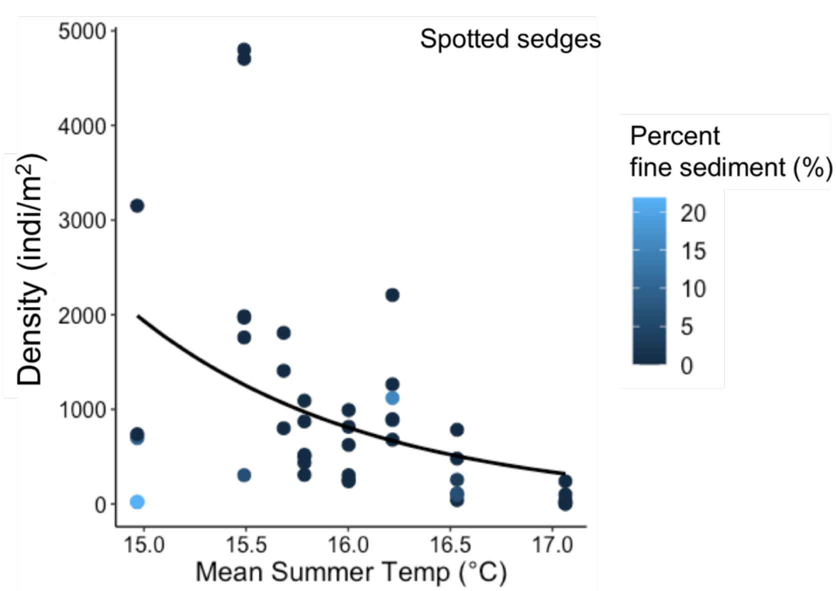

Fig. 10: Scatter plot of the effect of mean summer temperature and percent fine sediment on densities of spotted sedges.

Environmental factors

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

Across all sites, the upper Snake River had mean and mean maximum monthly summer temperature of 60.7°F and 67.0°F, respectively (Fig. 11). On average, mean temperatures *cooled* from the upper reaches (Moran – 62.7ºC), likely due to inflows of cooler water from undammed tributaries and shading provided by the canyon in the downstream reaches. Maximum mean monthly summer temperatures displayed a normal distribution, with the warmest temperatures occurring in the middle reaches (70.2 °F), between Moose and South Park (sites 3-5). Temperatures on the upper Snake are quite cool and are well-below the in-stream thermal tolerance limits of most aquatic 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 was also low across sampling sites. Percent dewatering ranged from -3% to 17%, with the highest values at Deadman's Bar and Pritchard Boat Ramp. Dewatering levels are significantly lower than recent measurements made on the Big Hole River, Montana (Birrell et al. 2022) and suggest that little anthropogenic water withdraws occur during summer. Moderate dewatering at some sites may be a result of recent, locally low snowpack levels of tributary watersheds or a degree of agricultural or municipal water withdrawals. Overall, dewatering levels do not appear to be of immediate conservation concern, as long as water conservation policies remain intact and sufficient flows are released from Jackson Lake Dam. However, previously proposed cuts in discharge from Jackson Lake Dam could threaten the fishery. In spring, 2023, flows were proposed to be cut in 2023 from 280 ft^3/s – the state-recommended minimum flows for maintaining trout populations – to 50 ft^3/s (Ball 2023). While these cuts were eventually avoided, we urge water managers to avoid future reductions in flows, as these will also harm aquatic insect populations via decrements in oxygen availability and habitat space and possible increases in sedimentation. According to our models, if the above flow reductions were maintained through the summer, levels of dewatering on the upper river would exceed 95%.

Fine sediment levels were generally low, with values ranging from 0% to 17% across all samples. Overall, fine sediment made up an average of 2.2% of the substrate sampled, with no fine sediment detected at Moran, Moose, Wilson Bridge, and Pritchard (sites 1,3,4, and 6, respectively) and with the highest amount of fine sediment occurring at Sheep Gulch (9.4%; site 8). 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 upper Snake, assuming sediment levels are constant through time (Relyea et al. 2000). Sediment levels can, however, be highly variable over seasonal and annual scales, and a more consistent seasonal sampling of fine sediment may be warranted in the future.

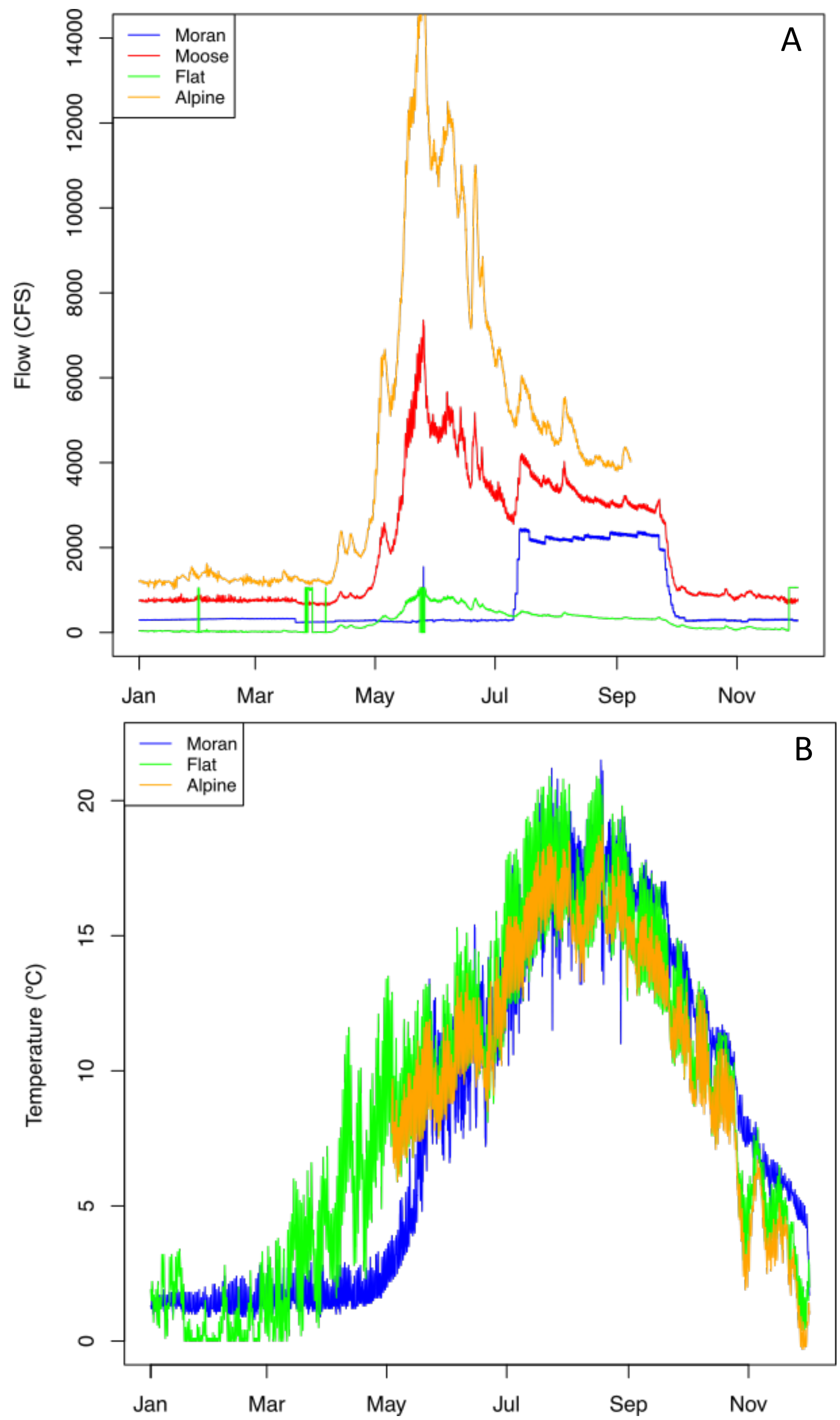

Fig. 11: Line plots showing river flows (ft[3]/s) for the year of 2023 at USGS gage stations at Moran, Moose, Flat Creek, and Alpine (A) and water temperature (°C) for the year of 2023 at USGS gage stations at Moran, Flat Creek, and Alpine. Temperature data for 2023 was not reliable at the Moose USGS station.

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

Habitat Condtion	Site 1: Moran	Site 2: Deadman's	Site 3: Moose	Site 4: Wilson Bridge	Site 5: South Park	Site 6: Pritchard	Site 7: Elbow	Site 8: Sheep Gulch
Mean Summer Temperature (°C)	17.1	16.5	15.7	16.0	16.2	15.8	15.5	15.0
Mean Maximum Monthly Summer Temperature (°C)	17.4	18.9	21.2	21.2	21.2	19.8	18.8	17.0
Mean Summer Temperature (°F)	62.7	61.8	60.2	60.8	61.2	60.4	59.9	58.9
Mean Maximum Monthly Summer Temperature (°F)	63.4	66.0	70.2	70.2	70.2	67.6	65.8	62.6
Mean Summer Flow Rate (cfs)	2527.4	2940.0	3600.2	4057.1	4365.4	4654.9	4839.3	5166.2
NHDPlus Modelled Summer Flow Rate (cfs)	2603.0	3560.9	3827.9	4025.1	4239.0	5464.1	5491.5	5958.7
Percent Dewatering (%)	2.90%	17.43%	5.95%	-0.80%	-2.98%	14.81%	11.88%	13.30%
Mean percent fine sediment (%)	0.0%	2.6%	0.0%	0.0%	4.2%	0.0%	1.0%	9.4%

Conclusion:

While continuing this monitoring program into the future will be key to confidently establishing 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 upper Snake 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. Previously proposed reductions in spring-time flows from Jackson Lake Dam (Ball 2023) would likely exacerbate fine sediment challenges by slowing flows in riffles, leading to increased sediment deposition, along with reductions in oxygen availability and habitat space. We encourage water and resource managers to avoid future flow reductions and other land use changes that could increase sediment loading or other environmental challenges. 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 also investigate patterns in nearby tributaries with more obvious human pressure or should incorporate additional environmental factors into the analysis like water chemistry and nutrients pollution.

Acknowledgements and data availability:

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Supplemental Tables:

Table S1: Least* salmonfly — *P. badia

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Mean Summer Temp.	2.97	0.001	1677	<0.001
Percent Fine Sediment	8.97	0.001	-5080	<0.001
AIC: 806.41; Pseudo-R^2: 0.515				

Table S2: Golden stonefly – H. pacifica

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Percent Fine Sediment	16.27	1.55	-10.49	< 0.001
AIC: 648.96; Pseudo-R^2: 0.154				

Table S3: Nocturnal stonefly — Classenia sabulosa

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Mean Max Monthly Summer Temp.	0.792	0.170	4.666	<0.001
AIC: 669.12; Pseudo-R^2: 0.763				

Table S4: Skwala – S. americana

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Mean Max Monthly Summer Temp.	-0.130	0.030	-4.405	<0.001
AIC: 1138.20; Pseudo-R^2: 0.474				
Model 2				
Predictor	Estimate	Std. Error	z-value	P-value
Percent Dewatering	2.413	0.873	2.765	0.006
AIC: 1142.36; Pseudo-R^2: 0.351				

Table S5: Green drake – D. grandis

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Mean Summer Temp.	2.97	0.001	1677	<0.001
Percent Fine Sediment	8.97	0.001	-5080	<0.001
AIC: 806.41; Pseudo-R^2: 0.515				

Table S6: Mahogany Dun — Paraleptophlebia spp.

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Percent Fine Sediment	13.113	3.356	3.907	<0.001

AIC: 306.46; Pseudo-R^2: 0.086

Table S7: March browns – Rhithrogena spp.

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Percent Fine Sediment	-21.6812	1.153	-18.8	<0.001

AIC: 2232.67; Pseudo-R^2: 0.429

Table S8: **Mother's day caddis — *Brachycentridae***

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Mean Summer Max Temp	1.142	0.325	3.516	<0.001
Percent Fine Sediment	- 1.085	0.123	- 8.836	<0.001

AIC: 30084.46; Pseudo-R2: 0.613

Table S9: Spotted sedges — Hydropsychidae

Model 1				
Predictor	Estimate	Std. Error	z-value	P-value
Mean Summer Temp.	-1.533	0.354	-4.337	<0.001
Percent Fine Sediment	-10.628	0.171	62.122	<0.001

AIC: 13657.05; Pseudo-R^2: 0.703