Memorandum

Date

|  |  |
| --- | --- |
| TO: | ClientFirstLastName, ClientCompany |
| FROM: | FirstLastName, Four Peaks Environmental Science & Data Solutions |
| SUBJECT: | Technical Memo Template |
|  |  |

Abstract

Use Heading 1 for introduction and any main sections of the memo

Introduction

Climate change poses numerous threats to aquatic ecosystems in the 21st century. Mean surface temperatures during the 2010s were 1.09 degrees Celsius than they were in the late 19th century (IPCC 2021). Precipitation patterns in the Pacific Northwest are also changing due to climate change. High magnitude winter storms have decreased in frequency in the Pacific Northwest between 1930 and 2007 (Kluver and Leathers 2015). Additionally, summer precipitation and stream flows are also expected to be reduced in the region, effects that are likely to generate increased water temperatures (Crozier et al 2021; Hamlet and Lettenmaier 1999; Mote et al. 2003). While increased water temperatures can impact both juvenile and adult salmonid life stages directly through increased risks of disease and mortality (Crozier and Siegel 2017; Bailey 2017; Martin et al. 2017), changing water temperatures also have the potential to generate cascading effects on the benthic macroinvertebrate forage base (Hinz and Wiley 1998; Kishi et al. 2005). Here we examine the potential impacts of increased water temperatures to stream macroinvertebrate communities and the ensuing potential impacts to Pacific Northwest Salmonids.

Methods

Use Heading 1 for introduction and any main sections of the memo

Results

Based on studies performed around the world, there are several key ecological trends generally observed in aquatic benthic macroinvertebrate (BMI) communities that are generally associated with increasing temperature. One of the predominant themes identified is that taxa adapted to colder conditions tender to move towards higher latitudes and are replaced with species more tolerant of warm conditions (Daufresne et al. 2007; Durance and Omerond 2007; Lento et al. 2022). At the local level, this is reflected in changes in BMI community compositions, which has been reflected in numerous studies worldwide.

**Trends in taxa richness**

In a study of midwestern streams in Wisconsin and Minnesota, Wang and Kanehl (2003) found increases in multiple temperature metrics including maximum 7-day temperature and minimum temperature during the first 3 weeks of July were associated with increasing taxon richness. Similar results were found by Castella et al. (2001) who examined European glacial fed streams from Svalbard to the Pyrenees, and found increasing maximum stream temperature was associated with increasing richness and a shift in density and presence in some taxa. Culp et al. (2019) used the average of maximum august air temperatures observed between 1971 and 2000 an index of temperature experienced by arctic BMI communities. They found that higher latitudes (with colder air temperatures) had reduced taxonomic richness (Culp et al. 2019), with other arctic studies identifying similar trends (Lento et al. 2022). Air temperature has been demonstrated to be a reasonable indicator of water temp (Yang et al. 2021). Similar trends in reduced taxa richness at higher latitudes and elevations (that typically experience colder temperatures) have been observed in comparisons between sites in Ecuador and Denmark (Jacobsen et al. 1997). Even within the same region (thus controlling for large scale changes in latitude), temperature can strongly influence BMI community composition. Phillips et al (2015) compared the BMI community of a large Canadian river system artificially cooled in the summer by a hypolimnetic release dam to non-thermally impacted reference reaches and found reduced richness and diversity compared to the reference sites.

The trend of decreasing richness at higher latitudes and in colder environments may be explained by the physiological tolerance hypothesis which posits that more physiological traits can live in warm/wet conditions than cold/dry Currie et al. (2004). However, the study sites of Castella et al. (2001) all had max temperatures below 16 Celsius and many taxa (though not all, possibly due to not reaching the upper thermal limits of those taxa) densities were relatively hump shaped, reflecting preference for a thermal optima (Castella et al. 2001). This indicates trends in community metrics associated with temperature may not be linear, and may depend on the range of temperatures measured, as most species have a thermal optimal above and below which their persistence will be hindered (Vannote and Sweeney 1980). Thus, the findings of increasing richness with increasing maximum stream temperature by Castella et al. (2001) are best applied in the context of cold glacial streams. However, other studies have even found opposite trends in BMI richness with temperature. Arai et al. (2014) observed strong negative associations between June water temperatures and taxon richness in Japanese headwater streams.

Additionally, multiple studies have also found no clear trend in richness with temperature. Mustonen et al (2018) modeled future expected temperatures at these Finnish stream sites with current (1998-2010) BMI community data for the end of the 21st century and evaluated changes in predicted community structure and richness using the River Invertebrate Prediction and Classification System multitaxon distribution model. Trends in richness were variable depending on initial temperature changing the most at sites with an initial MAAT below -2 Celsius and the least at sites with an initial MAAT near 0 Celsius. However, they found that community assemblage (measured with Bray Curtis CITE) changed most for streams experiencing the coldest initial (1981 to 2010) conditions as measured by mean annual air temperature (MAAT; Mustonen et al (2018). These cold high latitude sites also generally experienced the greatest projected increases in temperature. Thus, they concluded that changes in BMI community taxa structure may be more substantial than changes in richness in many streams experiencing warming conditions as species tolerant of warm conditions replace cold-water species that become locally extirpated (Mustonen et al 2018). In their study of Northern California BMI communities from a 20 year dataset, Lawrence et al. (2010) also found that increasing degree days were associated with changes in community composition from those typical of cool wet years to those typical of warm dry years but no changes in taxa richness.

Thus, the varying results of these studies assessing relationships between taxa richness and temperature indicate that assessing taxa richness alone may not always be a good indicator of impacts to BMI communities from climate change. This may be because taxa richness is sensitive to a myriad of factors including latitude, local environmental conditions (such as stream acidity; Bradley & Ormerod, 2002; Kowalik & Ormerod 2006), and the range of temperatures measured.

Community composition (in terms of relative abundances or biomasses of certain taxa, or functional feeding groups (FFGs) represent alternative ways to assess changes in ecosystems. Functional groups represent coarse groupings of BMI taxa based on their diet and role in the ecosystem. Shredders are typically found in small headwater streams with high allochthonous leaf input as their primary food source, and serve to break down this coarse particulate organic matter (CPOM) into fine particulate organic matter (FPOM) (Vannote et al. 1980). As the stream widens downstream with input from tributaries and becomes more exposed to sunlight, primary production serves as a food source for grazers, and collectors make use of the FPOM generated from upstream Vannote et al. (1980). Turbid high order rivers have little primary productivity and are generally dominated by collectors Vannote et al. (1980). Predator abundances typically remain relatively constant through the reaches of a watershed Vannote et al. (1980). Thus, in addition to illuminating changes in community structure, functional feeding groups allow the direct and indirect processes resulting in the changes to be identified.

**Trends In Community Composition**

*Functional Feeding Groups*

Hinz and Wiley (1998) found that in Northern Michigan cold-water streams, water temperature had a substantial impact on BMI functional feeding groups and trout growth rates, though different temperature metrics were better predictors of biomass for different functional groups. July temperature summary metrics (e.g., mean, maximum, minimum) were significantly positively associated with total BMI filter feeder, and grazer densities. Grazers, filter feeders, and predators were all significantly positively associated with summer temperature fluctuations (Hinz and Wiley 1998). Annual maximum temperature and average yearly daily temperature fluctuation were positively associated with total biomass (Hinz and Wiley 1998). Other temperature metrics were not consistently associated with temperature across functional feeding groups (Hinz and Wiley 1998). Hinz and Wiley (1998) also found significant positive associations between temperature and juvenile brook trout growth rates, though half of the temperature effect were indirect effects of temperature on growth through temperature impacts on BMI biomass (Hinz and Wiley 1998). It should be noted that this is another example of a system with very cold water temperatures (maximum average daily temperature of 14.4 Celsius) and thus likely represents systems on the lower side of the physiological optima (Hinz and Wiley 1998). Trends may be different if temperatures were substantially warmed.

Wang and Kanehl (2003) performed a study on the BMI communities of streams in Minnesota and Wisconsin, also paying particular attention to functional feeding group compositions. Using Canonical Correspondence Analysis (CITE), they found that the maximum 7 day average temperature (which ranged between 13.5 and 27.2 Celsius) and the minimum temperature during the first three weeks of July (which ranged between 11.8 and 26.0 Celsius) were important predictors of BMI community composition (Wang and Kanehl 2003). The percentage of the BMI community represented by filterers, gatherers, and EPT individuals decreased as temperature increased (Wang and Kanehl 2003).

While temperature clearly plays a role in determining the composition of functional feeding groups, because the riparian landscape impacts allochthonous food inputs, the riparian landscape also exhibits an influence (Houghton et al. 2002). Additionally, functional feeding group community composition changes seasonally (Fierro et al. 2015; Hawkins and Sedell. 1981) so BMI sampling done sporadically and at different times of year may give a misleading impression of community changes. For example, in a study of Oregon streams, Hawkins and Sedell. (1981) found that shredder abundances were substantially greater in fall, possibly linked to increased allochthonous input of coarse particulate organic matter.

*Taxa Composition*

Besides the relatively coarse grouping of functional feeding groups, communities can also be grouped by abundances or biomass of specific taxa. In a study of BMI communities in streams in southeastern New Brunswick, Canada, Lento et al. (2020) observed that the stoneflie families Pteronarcyidae, Perlidae, and Perlodidae were more common at sites with a greater number of June through August degree days above 19 Celsius, whereas Leuctridae, Nemouridae, and Chloroperlidae were associated with cooler sites. Temperature has also been shown to influence BMI community taxon composition in Chilean streams Fierro et al. (2019), large European rivers like the Rhone (Daufresne et al. 2003, 2007), Canadian subarctic lakes Pretty et al. (2021), and New Zealand streams Quinn and Hickey (1990). In the Pacific Northwest, studies have demonstrated associations between community compositions and a variety of different temperature metrics in Columbia Basin streams in Idaho and Oregon (Ott and Maret 2001; Waite et al. 2020), Puget Sound streams in Washington (Waite et al. 2020).

Durance and Omerond (2007) found similar trends in BMI community composition with warming temperatures in UK streams. They found warming winter stream temperatures had moderate impacts to community structure (measured with Detrended Correspondence Analysis; Van Der Maarel 1969) and decreased April BMI abundances (Durance and Omerond 2007). However, these trends only occurred in streams with relatively neutral pHs and not in more acidic streams which already had reduced BMI community richness and abundance, indicating an overriding impact of acidification that should be taken into account (Durance and Omerond 2007; Bradley & Ormerod 2002; Kowalik & Ormerod 2006). The reductions in April abundance are likely due to modified emergence timings associated with faster growth (Elliott et la. 1988) or changing energy flows (Kishi et al. 2005; Lepori et al. 2005).

The scale of taxonomic resolution used can impact the results of a study and should be considered carefully. Coarser resolutions save time and cost but may obscure important trends identifiable only in fine scale taxonomic resolutions (Jones 2008). For example, in a study of BMI communities in New Zealand Rivers Quinn and Hickey (1990) found Plecoptera were restricted to cooler streams (<13 C mean annual daytime temp) and in a study of Japanese headwater streams Arai et al (2014) found that maximum water temperature was strongly negatively correlated with Plecoptera density. However, Lento et al. (2020) demonstrated that temperature preferences within the Plecoptera order can vary even among families and so identification to order may not be sufficient.

The exact taxonomic resolution needed to adequately identify temperature regimes, however, remains unclear. Bowman and Bailey (1997) found that genus and family order taxonomic resolutions yielded similar assessments of community structures. However, Hubler et al. (2024) found that when assessing the influence of maximum weekly maximum temperature (MWMT) on community composition, finer scale taxonomic resolutions best distinguished different thermal regimes (Hubler et al. 2024). They found that even within multi-specific genera, many species had different thermal optima (Hubler et al. 2024). They also observed that the most thermally sensitive or tolerant BMI taxa were the best indicators of differing thermal regimes (Hubler et al. 2024). Therefore, if the goal of a BMI assessment is to identify changes in temperature regimes, focusing on identifying the most thermally tolerant or sensitive taxa at a fine taxonomic resolution may be the best use of limited time and financial resources.

**Life History Changes**

As alluded to previously, changes in key life history traits are expected to change in BMI communities with changes in temperature associated with climate change. This is because an organism’s consumption, growth, fecundity, and energy use can be fundamentally linked to metabolism, which itself is dependent on temperature and body size (Brown et al. 2004). Indeed, metabolism increases with temperature, and metabolism per unit of body weight decreases with increasing body mass (Brown et al. 2004). This manifests for ectotherms in the temperature size rule which hypothesizes decreasing body size with increasing temperature (Daufresne et al. 2009). In insects temperature cues growth, development, and emergence and so higher temperatures are associated with increased development, maturation rates, increased metabolic expenditures and reduced size at maturity and generation times (Ward and Stanford 1982; Vannote and Sweeney 1980).

The above hypothesis for how temperature and bioenergetics interact to inform changes in ecosystems have been supported through numerous field studies. Vannote and Sweeney (1980) demonstrated how temperature can impact growth, metabolism, fecundity, and generation time, and size at emergence of aquatic insects as temperatures move away from the thermal optima. Vannote and Sweeney (1980) hypothesized that movement away from the thermal optima were associated with smaller, less fecund individuals. In warmer than optimal environments, metabolic rates are higher (Brown et al. 2024) and individuals consequently tend to mature and develop adult features quickly (short generation times) at small sizes and thus demonstrate low fecundity (Vannote and Sweeney 1980). In colder than optimal environments growth rates are slower and consequently populations tend to have long generation times which also produces smaller sizes at maturity and lower fecundity than under an optimal thermal regime (Vannote and Sweeney 1980). Because of these characteristic changes in growth and generation times, Vannote and Sweeney (1980) observed in field and experimental studies that warmer temperatures were associated with earlier emergence, reduced adult body size, and lower fecundity. The meta analysis of Daufresne et al. (2009) supported these fundamental energetic hypotheses for ectotherms, demonstrating an increase in relative abundances of smaller species, younger age classes and reduced size at age with increasing temperature, consistent with the temperature size.

Vannote and Sweeney (1980) additionally noted that adult emergence was synchronous and increased temperatures accelerated adult tissue growth and thus initiated emergence independently of size. Concurrent with their hypothesis, they observed that for species with many generations within a year, summer and winter emerging cohorts were different in size (with summer cohorts being smaller; Vannote and Sweeney 1980). The varying thermal optima of different BMI taxa consequent leads to temporal separation of time periods of juvenile rearing and emergence (Vannote et al. 1980).

**Impacts to Insectivorous Fish**

Changes in BMI communities, whether in terms of taxa richness, abundance, or community composition can have cascading impacts to fish communities that use invertebrates as a food source. For example, using a causal path model Hinz and Wiley (1998) found that while temperature most affected juvenile brook trout growth rates, half of the temperature effect was an indirect effect of temperature on growth. This occurred through temperature impacts to the invertebrate food source ratio, with the biomass of predator and grazer invertebrates increasing with increasing temperatures (Hinz and Wiley (1998). Thus, temperature exerts direct influences on fish growth, but also indirect through influencing prey availability (BMI biomass). The combined roles of temperature and prey availability in influencing fish growth is important, and in food limited environments full growth potential may not be realized (Weber et al. 2014). Brett (1982) found juvenile Chinook salmon fed to satiation in a laboratory experienced maximized growth at 19 degrees Celsius, but that natural source populations fed about 60% of maximum, with an estimated corresponding temperature for optimal growth of 14.8 Celsius. When temperatures are near their thermal optima, fish can exert a strong top down effect on BMI abundances through predation (Kishi et al. 2005).

Environments with an abundant food source of aquatic invertebrates can offset the negative impacts to salmonid growth associated with increased temperatures (and increased metabolism) from climate change, allowing salmonid populations to persist at temperatures above their thermal optima (Lusardi et al. 2020; Swartz and Warren 2022). However, the extent of this offset may vary with lifestage (Swartz and Warren 2022). Swartz and Warren 2022 found juvenile cutthroat trout benefited in terms of biomass, density, and average size when experiencing elevated stream temperatures following a forest fire, however, adults did not experience these positive affects. This may be related to the temperature size rule which posits and increase in smaller age classes with warmer temperatures (Daufresne et al. 2009). These smaller individuals will have less total metabolic expenditures than large individuals.

**Discussion**

**Can we format discussion to directly answer Lucius’s questions but in a way that flows. Synthesizing the above information.**

* 1. How do different changes in temperature correspond to changes in emergence timing versus overall community structure?
  + That is, do small changes result in changes in timing only? Or, can even small changes result in changes in community structure?
    - Changes in temperature influence changes in community structure (in terms of both functional feeding groups and taxa abundances) both seasonally (Durance and Omerond 2007; Fierro et al. 2015; Hawkins and Sedell. 1981; Vannote and Sweeney 1980), and as a response to longer term temperature trends (Fierro et al. 2019; Lento et al. 2020; Daufresne et al. 2003, 2007; Quinn and Hickey 1990; Ott and Maret 2001; Waite et al. 2020; Wang and Kanehl 2003; Hinz and Wiley 1998).
    - Changes in emergence timing inherently change the composition of the BMI community, as even under natural conditions the different life stages of different taxa tend to be naturally temporally separated by their thermal optima (Durance and Omerond 2007; Vannote and Sweeney 1980). Thus, it is difficult to disentangle changes in the timing of key life history events from changes in abundance when looking at larval stages as these phenomena are inherently linked.
    - In short, emergence timing, body size, and fecundity for a particular taxa change along a temperature gradient in relation to their thermal optima, and these natural separations in thermal optima tend to lead to a gradient of community compositions.
  + Is there a tipping point above which changes in T result in changes in community structure, but below which the community remains intact but the taxa present emerge at different times?
* 2. What aspects of temperature (e.g., averages, threshold exceedances, maxima) have the most impact on BMI communities? 3. What metrics (e.g., annual mean, annual max, annual max 7DADA, annual ATUs, mean or max or 7DADA/MWAT during critical periods) best predict BMI community shifts?
* 3. What metrics (e.g., annual mean, annual max, annual max 7DADA, annual ATUs, mean or max or 7DADA/MWAT during critical periods) best predict BMI community shifts?

Temperature metrics indicative of the magnitude and duration of thermal stress tend to be the best indicators of BMI community composition and species presence, and thus appear to have the most impact on BMI communities. Such metrics of thermal stress include various summary metrics (e.g., minimum, maximum, mean) during months of highest temperatures (Hinz and Wiley 1998; Arai et al. 2014), maximum daily (or 7 day) average temperatures, maximum daily (or 7 day) maximum temperature, and metrics of temperature accumulation such as degree days (Hubler et al. 2024; Phillips et al 2015; Ott and Maret 2003; Wang and Kanehl).

Ott and maret is a great source where they tested 9 different temperature metrics.

* 4. What community metrics (e.g., FFG composition, taxa richness, total abundance, taxa specific abundance, BIBI scores) best represent shifts that matter for anadromous salmonids?

The most BMI community metrics most relevant to salmonids appear to be metrics of community composition and abundance. One of the key habitat features for coldwater species like salmonids is thermal suitability. To identify thermally suitable habitat, assessments of community composition, paying particular abundance to the abundances and presence/absence of expected coldwater taxa are likely to be of greatest value. Identifying the presence or absence of expected coldwater taxa is important given that thermal preferences within orders vary (Lento et al. 2020; Hubler et al. 2020). However, the suite of temperatures over which salmonids can survive and grow is influenced by food availability (Lusardi et al. 2020; Swartz and Warren 2022), with food availability itself being influenced by temperature (Hinz and Wiley 1998). Food availability has been demonstrated to play a strong role in determining salmonid thermal optima (Brett 1982). Thus, metrics of BMI abundance and density are important to assess, particularly in habitats that often experience temperatures stressful to salmonids as food availability may determine their ability to persist in these environments.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Study | Temp Metric 1 | Temp Metric 2 | Temp metric 3 | Eco Metric 1 | Eco Metric 2 | Eco metric 3 |
| Arai et al. 2015 | Max water temp | July and August temperature (1 sample each month) | Max water temp | Plecoptera density (negative) | Plecoptera density (negative) | Richness (negative (higher temps lower richness)) |
| Castella et al. | Max water temperature |  |  |  |  | Richness (positive association (higher temps higher richness)) |
| Culp et al. | average maximum August AIR temperatures |  |  |  |  | Richness (positive) |
| Daufresne et al. 2007 | Normalized mean yearly water temperature |  |  | Observed gradual changes Community structure with climate change |  |  |
| Durance and Ormerod 2007 | Mean modeled winter stream temperature (December-March) |  |  | Community composition via Detrended Correspondence Analysis | Total abundance (negative) total abundance decreased an average of 21% for each 1 Celsius increase in temperature | Richness was not correlate with stream temperature |
| Fierro et al 2015 | Instantaneous water temperature |  |  | Order and FFG composition via bray curtis similarity index | FFG relative abundance | Found taxa richness was greater in summer than winter |
| Fierro et al. 2019 | Instantaneous water temperature |  |  | taxa richness; Ephemeroptera, Plecoptera, and Trichoptera richness; ShannonWeaver diversity; and abundances | RDA and CCA for community composition |  |
| Hawkins and Sedell 1981 |  |  |  | Numerical and relative abundances of FFGs at different river reaches (longitudinal) and in different seasons |  |  |
| Hinz and Wiley | Annual max temp, average annual daily temp flux | Summer temperature flux | July temp flux | biomass | Grazer biomass | Grazer biomass |
| Hinz and Wiley | Summer temp summaries | July temp summaries | Annual maximum and temp fluxes | Filter feeder biomass | Filter feeder biomass | Filter feeder biomass |
| Houghton et al 2022 |  |  |  |  |  |  |
| Hubler et al. 2024 | Modeled stream temperature Maximum weekly maximum temperature (MWMT) |  |  | Taxa specific probability of occurrence. Was able to assign taxa to thermal preferences. |  |  |
| Jacobsen et al | Maximum stream temperature |  |  | Taxa richness (positive association) |  |  |
| Lawrence et al | Degree days (10 celsius baseline) |  |  | No association between Ratio of expected to observed taxa, %ept and richness |  |  |
| Lento et al. 2022 | long term average maximum august air temperature from 1970 to 2000 |  |  | Rarefied alpha diversity (taxa RICHNESS rarefied to same number of sample sites within catchment) |  |  |
| Lento et al. (2020) | Degree days above 19 Celsius |  |  | Used Redundancy analysis (RDA) to link environmental variables including degree days to identify abiotic factors most influencing BMI composition |  |  |
| Mustonnen et al | Mean Annual Air Temperature MAAT (MAAT) |  |  | Community dissimilarity via Bray Curtis |  |  |
| Phillips | Degree days (10 Celsius threshold) | Mean weekly water temperature |  | Assessed if minimum degree days was enough for different mayfly species | More of a qualitative assessment, but taxa **richness and diversity were lower below the dam which experience colder summer water temperatures** | Community structure assessed with NMDS |
| Pretty et al. | Instantaneous water temperature |  |  | Community assemblage via redundancy analysis |  |  |
| Quinn and Hickey | Mean annual temperature (MANT) | MANT + half mean winter—summer range (°C) |  | Stonefly biomass | Stonefly biomass and Plecoptera biomass. |  |
| Tate and Heiny | Instantaneous water temp |  |  | Detrended correspondence analyses. Regressed specific environmental variables with DCA axes. Temperature explained 50% of variability in DCA axis 1 | Shannon diversity |  |
| Vannote and Sweeney 1980 | Degree days | Water temperature |  |  |  |  |
| Waite et al. 2020 | Maximum minimum water temperature for the 42 days before sampling |  |  | Gradient Forest |  |  |
| Wang and Kanehl | Maximum of the 7 day mean temperature | minimum of first three weeks of July temperature |  | CCA | Relative abundance FFG | Richness (number of taxa) positively associated with temperature metrics |
| Ott and Maret | Compared 9 temperature metrics |  |  | Correlations were all over 0.70 | **Coldwater taxa negatively associated with all temperature metrics. Best metric taxa depended.** |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |

* What community metrics (e.g., FFG composition, taxa richness, total abundance, taxa specific abundance, BIBI scores) best represent shifts that matter for anadromous salmonids?

References

Four Peaks Environmental Science & Data Solutions, 2017. Report Title, Final Report submitted to Client Name, Month.

Weiland, M., Miller, J.L., Murauskas, J.G. and Haffey, S.F., 2017. Journal article title. Journal Name, Vol. XX(Issue No.): pp-pp