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Susitna-Watana Hydroelectric Project
(FERC No. 14241)

River Productivity Study (Study 9.8)

Appendix A
Review of the Effects of Hydropower on Factors Controlling Benthic Communities

Initial Study Report

Prepared for

Alaska Energy Authority

SUSITNA-WATANA HYDRO

Clean, reliable energy for the next 100 years.

Prepared by

R2 Resource Consultants, Inc.

February 2014 Draft

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LIST OF ACRONYMS AND SCIENTIFIC LABELS

Abbreviation	Definition
ADF&G	Alaska Department of Fish and Game
AEA	Alaska Energy Authority
APA	Alaska Power Authority
BMI	benthic macroinvertebrates
C	Carbon
CPOM	course particulate organic matter
DNA	Deoxyribonucleic acid
DOC	Dissolved organic carbon
DOM	Dissolved organic matter
EPT	Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)
FERC	Federal Energy Regulatory Commission
FPC	Flood Pulse Concept
FPOM	fine benthic organic matter
GIS	Geographic Information System
HSC	Habitat Suitability Criteria
ILP	Integrated Licensing Process
N	Nitrogen
PAR	Photosynthetically Active Radiation
PM&E	protection, mitigation and enhancement
POC	Particulate organic carbon
POM	Particulate organic matter
PRM	Project River Mile
PTF	Pulse Type Flow
RCC	River Continuum Concept
RM	River Mile(s) referencing those of the 1980s APA Project.
SDC	Serial Discontinuity Concept
WUA	Weighted Usable Area

EXECUTIVE SUMMARY

This paper includes a review of over 500 reports and published papers (Section 6) relating to the effects of river regulation upon benthic communities, with an emphasis on glacial rivers. The objective given in the Revised Study Plan Section 9.8 (AEA 2012) identified three general topics to be addressed by this paper. The first topic of this report was to provide a literature review summarizing relevant literature on macroinvertebrate and algal community information in Alaska, including 1980s Susitna River data reviewed. The second topic was to summarize literature on general influences of changes in flow, temperature, substrates, nutrients, organic matter, turbidity, light penetration, and riparian habitat on benthic communities. The final topic was to review and summarize the potential effects of dams and hydropower operations, including flushing flows and load-following, on benthic communities and their habitats. Each of these topics would be worthy of its own extensive literature review; therefore, to impart a general review of each within one report, is was necessary to present a brief overview, relating each topic to the others in order to synthesize a unified review that could be related to the Susitna-Watana Project.

Part of the consideration in writing this paper was to investigate the plausibility of a surrogate for the Susitna River. In addressing the first topic, reviewing information on Alaskan benthic communities, and specifically those reported as part of the the 1980s Susitna River studies, information was sparse, especially when considering examples of regulated rivers. Broadening the search to glacial river systems worldwide, the literature on benthic communities in glacial rivers largely came from studies conducted in the Swiss and Italian Alps and the French Pyrenees of Europe and in rivers throughout Sweden and Norway. Many of these papers examined the general ecology of glacial rivers, with only a handful of rivers actually regulated by dams or diversions. As for comparability to the Susitna Project, the European river systems with hydropower were found to be considerably smaller, often measuring less than 18.6 mi (30 km) in length, many within that distance to the glacier itself, with watershed catchments of 137.8 mi^2 (357 km^2) or less. For comparison, the Susitna River measures over 300 miles in total length, with the total basin covering 20,000 mi^2; at the USGS gaging station located at Gold Creek, Alaska, the drainage area measures 4,140 mi^2 (Kyle and Brabets 2001).

With a paucity of information on Alaskan or glacial-fed systems that would be comparable to the Susitna River, a reader's understanding of how the Susitna River ecosystem could be affected by hydroelectric operations would likely be based on well-known, peer-reviewed and grey literature studies examining the large river ecology concepts conducted in temperate rivers, and when considering the ecology of glacially-influenced river systems, many conceptual models originated from studies in those temperate alpine rivers in Europe, as well as Alaska. Thus, Section 2 contains a review of the River Continuum Concept and its corollaries, and the Flood Pulse Concept, as well as the conceptual models by Milner and others relating to glacial river ecosystems (Milner and Petts 1994; Petts et al. 2000; Milner et al. 2001; Uehlinger et al. 2002). With this general understanding of riverine ecology, the literature on benthic communities in Alaskan glacial systems was then reviewed, as well as information specific to the Susitna River collected during the previous licensing effort in the 1980s.

The review of large river ecology and glacial river ecology concepts also helped in addressing the second topic, to summarize literature on general influences of the key factors on the benthic communities in a large, glacially-fed river like the Susitna River. Large river ecology concepts revealed flow to be the most pervasive controlling factor, influencing other controlling factors such as temperature, substrates, the riparian vegetation, organic matter, and nutrients. Glacial river ecology stressed temperature and turbidity to be the most defining factors affecting the benthic communities in those systems. Each of these controlling factors was reviewed and summarized in Section 3 in the context of how they can affect BMI and algal communities.

The final topic was to review and summarize the potential effects of dams and hydropower operations, including flushing flows and load-following, on benthic communities and their habitats, addressed in Sections 4 and 5. In Section 4, this paper reviewed a specific class of operational effects most often associated with hydroelectric operations, pulse-type flows, or PTF. These PTF operations included power peaking, load following, flushing flows, recreation pulse flows, and others.

To conclude the paper, Section 5 addressed the effects of hydropower operations on the various controlling factors defined in Section 3, and the corresponding effect on benthic communities. Foremost in this discussion is the effect of altering the natural flow regime into one or more of the regulated flow regimes defined by Ward (1976): reduced flows, increased flows, seasonal flow constancy, and short-term flow fluctuations. The paper then summarized the consequences these flow modifications could have upon the benthic communities, and discussed the establishment of the varial zone, drift responses by BMI, and stranding during rapid flow reductions. Finally, the effects of river regulation on the other controlling factors were summarized, including the influence of impoundment of the reservoir on substrates and sediments, turbidity, and organic matter.

1. INTRODUCTION

The Alaska Energy Authority (AEA) is preparing a License Application that will be submitted to the Federal Energy Regulatory Commission (FERC) for the Susitna-Watana Hydroelectric Project (Project). The Project is located on the Susitna River, an approximately 300-mile-long river in Southcentral Alaska, which stretches from the Susitna Glacier to Cook Inlet. The Project's dam site would be located at Project RM (PRM) 187.1 of the updated Geographic Information System (GIS) based hydrography (Figure 1-1).

AEA filed the Revised Study Plan with FERC on December 14, 2012, which included a River Productivity Study (AEA 2012, Section 9.8). The overarching goal of the River Productivity Study is to collect baseline data to assist in evaluating the effects of Project-induced changes in flow and corresponding impacts to benthic macroinvertebrates (BMI), primary producers, and the lower-trophic lotic food web in the Middle and Lower Susitna River.

As described in the Study Plan (RSP Section 9.8.1., AEA 2012), the production of freshwater fishes in a given habitat is constrained both by the suitability of the abiotic environment and by the availability of food resources (Wipfli and Baxter 2010). Algae are an important base component in the lotic food web because they are responsible for the majority of photosynthesis in a river or stream, and serve as an important food source to many benthic macroinvertebrates (BMI). In turn, BMI are a linkage in flow of energy in the lotic food web as consumers that both regulate low-level organisms and are the primary food source for upper-level organisms, such as juvenile salmonids (Hynes 1970; Wallace and Webster 1996; Hershey and Lamberti 2001).

Benthic macroinvertebrates are also involved in the recycling of nutrients and the decomposition of terrestrial organic materials. Thus, they serve as a conduit for energy to flow from organic matter at the base of the food web to vertebrates at the top of the food web, e.g., freshwater salmonids (Hershey and Lamberti 2001; Hauer and Resh 1996; Reice and Wohlenberg 1993; Klemm et al. 1990). Marine-derived nutrients provided by adult anadromous salmon have the potential to increase freshwater and terrestrial ecosystem productivity (Wipfli et al. 1998; Cederholm et al. 1999; Chaloner and Wipfli 2002; Bilby et al. 2003; Hicks et al. 2005), and may subsidize otherwise nutrient-poor ecosystems (Cederholm et al. 1999). Recent studies have demonstrated the important role that BMI play in processing salmon carcasses in coastal streams (Cederholm et al. 1999, Chaloner and Wipfli 2002).

The functional roles that BMI and algae serve in food webs and energy flow in the freshwater ecosystem make these communities important elements in the study of a stream's ecology. The operations of the proposed Project may alter one or more environmental controlling factors that can affect the abundance and distribution of benthic algae and benthic macroinvertebrate populations which, in turn, may have an influence on fish growth and productivity. The degree of impact on benthic and fish communities resulting from hydropower operations will vary depending on the magnitude, frequency, duration, and timing of flows, as well as potential Project-related changes in temperature, substrates, nutrients, organic matter, turbidity, and riparian habitat. By investigating the current condition of algal populations, BMI, and food web interactions in the Susitna River, AEA will provide a better understanding of how changes in environmental factors might affect organisms at these trophic levels. Combining these baseline data with what is known about the effects of river regulation and hydropower operation, AEA can begin to assess the potential impacts of Project operations on bottom-up energy transfer and factors that affect river productivity in the Susitna River, as well as provide information for the development of any necessary protection, mitigation, and enhancement measures (PM&E).

Along the west coast of the United States, where anadromous salmonids are economically and culturally significant, large-scale mitigation programs have been funded (e.g., Bonneville Power Administration and CALFED), and these programs focus on assessment of impacts, and development of mitigation for flow regulation related to dam construction and operation, flow diversions, and river regulation. In addition, many studies have been conducted in conjunction with the relicensing of hydroelectric power projects, and these studies are partly designed to evaluate the effects of dam operations on downstream water quality and quantity, aquatic biota and habitats, channel structure and stability, and on recreational activities, such as rafting and fishing. This paper reviews an extensive hydropower literature base to help provide a basis of evaluating the future potential with effects in a glacially-fed Susitna River.

1.1. Objectives and Organization of This Paper

The effect of flow regulation on stream ecology and benthic communities has been widely studied (Ward 1976; Ward and Stanford 1979; Armitage 1984; Petts 1984; Cushman 1985; Saltveit et al. 1987; Dewson et al. 2007; Svendsen et al. 2009; Elosegi et al. 2010; Poff and Zimmerman 2010; Tockner et al. 2010). The objective of this paper is to present a synthesis of the current understanding of how hydroelectric operations can impact physical factors in the riverine environment, and how changes in those factors can alter the aquatic biological community and riparian habitat. Specifically, this review paper describes the natural effects of flow, temperature, substrates, turbidity, riparian habitat, nutrients, and organic matter on benthic algae and macroinvertebrates, and how Project operations might alter these physical factors, and in turn, affects the base of the food web. Algae and BMI are the focus of the review because they respond rapidly to changes in the environment or to presence of stressors, are likely the major source of energy for most fish species in the Susitna River, and because they represent two biotic components used to estimate river productivity.

2. OVERVIEW OF RIVER ECOLOGY AND GLACIAL RIVER CONCEPTS

The literature on aquatic and riparian ecosystems in glacial rivers of Alaska and other subarctic climates is limited. Much of the literature on benthic communities in glacial rivers comes from studies conducted in the temperate climate of Europe and Scandinavia. Thus, this paper begins by reviewing the best available scientific literature on river ecosystems, glacial river ecosystems, benthic communities in Alaskan glacial systems, and information specific to the Susitna River collected during the previous licensing effort in the 1980s. With this general understanding of riverine ecology in northern climates (Nearctic ecozone), factors affecting biological processes in glacial rivers, and Alaskan river benthic communities, this paper then reviews controlling factors that influence these communities (Section 3) and describes how hydropower project operations (Section 4) can affect river habitat and in turn, the corresponding benthic community (Section 5).

2.1. Large River Ecology Concepts

The river ecosystem is described by components that include: hydrology, diversity of channel and habitat types, solutes and sediments, and biota. While much of the understanding of lotic systems has been derived from investigations in small streams and lakes, several concepts have emerged that help explain large river ecosystems in form and function (Johnson et al. 1995; McCain 2013). Large-river ecosystems are highly dynamic, and gaining an improved understanding requires knowledge of a wide diversity of biotic and abiotic factors, ranging from life history and distribution patterns of stream biota, to production and energy flow, hydrodynamics, geomorphology, and the relationships among environmental and biotic factors (Johnson et al. 1995). Some of the most influential studies have combined the physical and biological aspects of river ecology into more holistic concepts that integrated the aquatic and terrestrial landscapes to help explain the complexity of large river ecosystems. Holistic approaches to river ecology studies account for interrelationships within and between the aquatic and terrestrial environments. Where possible, the influence of multiple environmental variables (multivariate approach) is related to biotic response that strengthens predictive model sensitivity. The primary theories important to the development of river ecology have been the River Continuum Concept and its corollaries, and the Flood Pulse Concept.

The River Continuum Concept or RCC (Vannote et al. 1980) is a heuristic tool that describes physical changes along the river continuum and how these features influence aquatic communities. Primary features that change along the river continuum that integrate stream order, energy sources, food webs, and, to a lesser extent, nutrients into a framework that explains the functional groups represented in each of the distinct zones along the continuum (Figure 2.1-1). The RCC predicts that for natural, unperturbed stream ecosystems there is a predictable longitudinal gradient of physical conditions that determines BMI, fish, and periphyton community structure and functions from headwaters to large downstream river segments. As hydrologic processes, food resources, nutrient dynamics, and riparian vegetation change with increasing stream size, the structural and functional composition of the macroinvertebrate community will change in a predictable fashion (Cummins 1979; Vannote et al. 1980; Sedell et al. 1989; Cross et al. 2013). For instance, in forested headwater streams, riparian vegetation limits autotrophic production through intense shading but also contributes large amounts of allochthonous material in the form of leaf litter, large woody debris and other organic matter (Figure 2.1-1). The RCC predicts the macroinvertebrate community will be dominated by invertebrate taxa that process coarse particulate organic matter, as well as collector invertebrate taxa that feed on the fine particulate organic matter produced by the feeding activity of shredders. Moving downstream, as stream size increases, the canopy opens and allows more light penetration to the stream bottom allowing primary productivity to increase, as does the contribution of grazing invertebrates to benthic community composition (Figure 2.1-1.).

Further downstream reaches of rivers receive substantial amounts of fine particle organic matter (FPOM) from upstream sources, and autotrophic production is often limited by depth and turbidity, leading to the dominance of collector invertebrate taxa. This concept applies well to northern temperate forested and confined river systems, and suggests that mid-sized rivers are relatively diverse compared to smaller and larger ones (Johnson et al. 1995; McCain 2013).

Corollaries to this framework include the nutrient resource spiraling concept (Newbold et al. 1981; Elwood et al. 1983) and the serial discontinuity concept (Ward and Stanford 1983a; Ward and Stanford 1995). The resource spiraling concept hypothesizes that resources (nutrients, carbon) do not flow downstream continuously, but instead are stored briefly within compartments (organisms, detritus, and waste materials), and are later released upon decomposition for continued biological recycling downstream (Elwood et al. 1983). Low-order streams have more organic matter retention and biological activity, so they tend to store resources, whereas larger rivers export resources more (Johnson et al. 1995). The serial discontinuity concept (SDC) (Ward and Stanford 1983a) describes the effects of river regulation on the hydrology and biological communities of rivers. The SDC recognizes that river regulation, and dams in particular, can disrupt the river continuum and effectively alter the stream ecosystem and macroinvertebrate communities downstream, thus creating a discontinuity. For instance, a dam on a large river would decrease turbidity downstream, increasing photosynthesis and diversity to more closely resemble a mid-sized river (Johnson et al. 1995). The SDC predicts that in response to the interruption of the river continuum, the stream system also has a tendency to return to natural or unregulated conditions as the distance from the dam or river regulation source increases (Ward and Stanford 1983a; Stanford and Ward 2001). The degree of change is dependent on the dam's location in the continuum, its operation, and the number of dams in a series on the river system.

Additional modifications to these ideas have sought to include lateral components of riverine ecosystems. The Flood Pulse Concept (FPC) described by Junk et al. (1989) recognizes that floods cause seasonal interactions between aquatic and terrestrial floodplain systems, which affect the riverine ecosystem (Tockner et al. 2000; Malard et al. 2006). In this framework, large rivers and their associated floodplains represent a single dynamic system created by strong linkages and interactions between hydrological and ecological processes (Tockner et al. 2000). Annual floodplain inundation results in exchanges of organic matter and other nutrients between river and floodplain environments, triggers blooms of primary and secondary production, and breeding and migration cycles of aquatic organisms. Although conceptualized as a model for large floodplain rivers in general, support for the FPC mainly comes from studies of unregulated temperate and tropical river systems where there is a regular seasonal pulse of floodwater inundating the floodplain and where water temperatures are relatively stable (Bayley 1995).

Contrasting with the RCC, the FPC concludes that large rivers with floodplains have the highest diversity (Johnson et al. 1995).

2.2. Glacial River Ecosystem Concepts

With consideration of the aforementioned river ecosystem concepts, headwater sources and climate can also have a profound influence on physical processes and biological dynamics in rivers (e.g., Beechie et al. 2006). Glacier-fed rivers have been defined and categorized by a number of researchers. In their classic paper on the ecology of glacial rivers, Milner and Petts (1994) described the classification of glacier-fed systems based on variations in temperature and the degree of influence of ice-melt, snow-melt and rain-dominated flows. This conceptual model of glacier-fed river types and the expected longitudinal distribution of river biota (Milner and Petts 1994) continue to be the base for understanding and predicting changes in invertebrate assemblages in these types of systems. Their model, based on a review and synthesis of both European and Alaskan literature, described the expected changes in zoobenthic communities as determined by two principle variables, water temperature and channel stability (Figure 2.2-1). Under this model, the immediate proglacial environment is characterized by temperatures ≤ 2°C and constantly shifting channels. This structured combination limits colonization with low temperatures, preventing successful colonization by taxa other than *Diamesa*. Channel instability also limits species colonization and abundances; highly unstable channels will lack invertebrates entirely. As distance from the glacier course increases and correspondingly temperature (>2°C to <4°C) and channel stability increases, other genera of Diamesinae may occur and together with representatives of Orthocladiinae, will dominate the community. The first mayfly and stonefly taxa to colonize glacial streams will typically be in the orders Baetidae (free-swimming), Nemouridae and Chloroperlidae (prefer coarse sand substrate interspersed among coarse gravel/cobble) and generally only occur once water temperature maxima exceed 4°C. As a more stable substratum develops in regions with temperatures above a daily maxima of 4°C representative Trichoptera will appear along with a more diverse fauna of ephemeropterans and plecopterans.As an extension to this model, Petts et al. (2000) incorporated the effects of seasonality in physical processes that in turn influence ecological processes in glacier-fed rivers. The authors proposed that glacial influence in rivers could be segregated into four phases that affect biological production:

- Phase 1 (late-winter to spring): short period of increased benthic productivity associated with nutrient rich, clear flowing water originating from snowmelt and increased groundwater input within stable channels;
- Phase 2 (late-spring to summer): the hydrograph rises with increased rates of glacial melt, flows become turbid, shear stresses increase, and bedload transport and instability increase. During this phase, the benthic community is constrained by the abrasive impact of mobile bed sediments, and increased turbidity further limits benthic algal growth (Uehlinger et al. 2002; Uehlinger et al. 2010);
- Phase 3 (late-summer to fall): the hydrograph recedes, bedload transport decreases, channels stabilize, groundwater inputs increase, and turbidity correspondingly decreases. Algal response to these improved conditions can be rapid, thus providing a food base for increased benthic invertebrate production throughout the fall, and growth in the winter

months (Füreder et al. 2001; Robinson et al. 2001; Schütz et al. 2001; Uehlinger et al. 2002; Uehlinger et al.2010);

- Phase 4 (late-fall through winter): comparatively stable and cool conditions under ice and snow that cover the surface of the river. The physical stability of this period may be critical for many biological processes and organisms especially in open-water reaches, which could have increased primary production that can benefit grazer taxa in the winter months, resulting in large abundances of benthic invertebrates in those areas (Füreder et al. 2001; Schütz et al. 2001).

These seasonal phases translate well to the concept of ecological "windows of opportunity" in glacial stream ecosystems described by Uehlinger et al. (2002; 2010). This concept states that there are distinct periods, in the spring and fall, in which environmental conditions are optimal for primary production, which in turn supports increased abundances of benthic invertebrates (Figure 2.2-2). These windows are brought about due to the seasonal changes in flow, which affect temperature, bed load, turbidity, and nutrients, in conjunction with light availability (Uehlinger et al. 2002; Uehlinger et al. 2010).

Milner et al. (2001) presented an update to the Milner and Petts (1994) model, based on field results from a large study that evaluated macroinvertebrate composition in glacially influenced rivers at different latitudes in Europe (Figure 2.2-3). The authors used a generalized additive statistical model to predict macroinvertebrate taxa richness at the reach level from 11 environmental variables. Their analysis indicated that maximum water temperature and channel stability explained the majority of variability (identified as the primary controlling factor) across all 40 taxa included in their model. Many taxa displayed a threshold temperature and stability rank, below which those taxa were absent. Significant breaks in community composition were shown for various groups at 2, 4 and 8°C (Figure 2.2-3), and at a value of approximately 40 on the Pfankuch Stability Index channel bottom score (Pfankuch 1975). This study also identified the disjunct nature of habitat change over large downstream gradients in glacial rivers. In contrast to earlier conceptual models developed in temperate systems (e.g., Vannote et al. 1980), biological communities in glacial systems did not necessarily follow downstream gradients because physical gradients could be profoundly altered by tributary and groundwater inputs, or other disturbances (Ward and Stanford 1983a; Sedell et al. 1989; Ward and Stanford 1995; Ward and Uehlinger 2003).

Passovoy (2002) applied the model from Milner and Petts (1994) to nine habitat types defined by Murphy et al. (1989), which included main channels, backwaters, braids, channel edges, and sloughs in the main river channel, and beaver ponds, terrace tributaries, tributary mouths, and upland sloughs across the valley floor. This application predicted that the model developed by Milner and Petts (1994) would chiefly apply to Diamesinae chironomids inhabiting main channels because of the greater exposure to disturbance. In contrast, sloughs and backwaters would have lower levels of short-term disturbances, and would have the same taxa as the main channel, but due to increased stability and lower velocities would also include Beatidae and Nemouridae or Chloroperlidae (Passovoy 2002). Thus, off channel habitats would provide greater environmental stability, and would therefore support a greater diversity of invertebrate taxa. The effect of tributaries would be variable, with effects on the invertebrate community depending on whether the tributary itself was glacial or non-glacial (Brittain and Milner 2001). Communities of invertebrates differ between mainstems and tributaries, and more diverse invertebrate drift communities from non-glacial tributaries could increase diversity in glacially-influenced mainstems (Saltveit et al. 2001). In combination, these papers describe a distinction between the effects of physical processes on invertebrate communities in main channel and off-channel habitats in glacial rivers.

Unfortunately, not all known controlling factors have been incorporated into glacial river ecosystem models. For example, winter ice formation in glacial rivers can be a major component affecting physical and biological processes (Scrimgeour et al. 1994; Prowse 2001b). Surficial river ice formation and breakup is a dominant factor whose dynamics vary spatially and temporally (Bieniek et al. 2011). Ice processes influence hydrologic, geomorphic, and chemical characteristics of a river, and likely have an enormous influence on the aquatic ecosystem in glacial rivers like the Susitna River (Prowse 2001a; Prowse 2001b; Bieniek et al. 2011). The topic of the biological, chemical, hydrologic, and geomorphological functions associated with river-ice ecology is too complex to completely review in this paper, but was thoroughly reviewed by Prowse (2001a; 2001b) and Scrimgeour et al. (1994). Nevertheless, the magnitude of this annual disturbance is important to consider in the context of this discussion on glacial rivers and Alaskan benthic communities, so will be briefly discussed here.

Ice forms on the surface of many rivers in Alaska during the late fall-early winter, and colder temperatures during mid-winter can cause some locations to freeze completely (Prowse 2001a; Prowse 2001b). Ice breakup is an annual disturbance, occurring during late winter-early spring (Scrimgeour et al. 1994; Prowse 2001a; Prowse 2001b). River ice breakups are characterized by large increases in current velocity, stage, water temperature, suspended sediment concentrations, and substrate scouring (Scrimgeour et al. 1994). Dynamic breakups cause a congestion of large pieces of ice to move downstream at reported velocities in excess of 5 m/s and increasing water levels at over 1 m/min (Scrimgeour et al. 1994; Prowse 2001b). The potential ecological effect of such a physical disturbance far surpasses anything that is possible except under rare and extreme open-water flow conditions, best paralleling a major flooding event, which can reduce benthic invertebrate and algal density, biomass, and taxonomic richness, depending on the magnitude (Scrimgeour et al. 1994).

2.3. Benthic Communities in Alaskan Glacial Rivers

Alaskan rivers generally exhibit lower macroinvertebrate species richness than temperate river systems (Oswood 1997) due in part to the limitation of colonization rates by low temperatures (Flory and Milner 2000). Both growth rate and diversity of aquatic fauna in these systems are typically low compared to temperate systems; life histories of aquatic invertebrates in Alaskan streams are extended, being typically univoltine (one generation per year) or semivoltine (one generation per two years) (Oswood 1989; Stewart et al. 1990). Diptera (true flies), Ephemeroptera (mayflies), Plecoptera (stoneflies) and Tricoptera (caddisflies) are considered representative orders in glacial rivers (Oswood 1989; Oswood 1997; Passovoy 2002); furthermore, the dominance of chironomid midges in glacial waters worldwide is well documented (Milner and Petts 1994; Brittain and Milner 2001; Snook and Milner 2001). Most of the characteristic midge species are within the genus *Diamesa* (Oswood 1989; Oswood 1997). Indeed, the most remarkable feature of Alaskan benthic fauna in comparison to temperate North American systems may be the dominance of Diptera, which constitute an average of 59-percent of the invertebrate fauna of Alaskan headwater streams (Oswood 1989). Although the relative taxonomic composition of Alaskan streams is impoverished in comparison to temperate North American streams, benthic invertebrates do not appear to have notably lower densities (Cowan and Oswood 1984). Spring-fed streams in Alaska are frequently characterized by remarkably high densities of Chironomidae and densities of these invertebrates in other stream types (including glacial source streams) may be high as well (Cowan and Oswood 1984; Oswood 1989; Oswood 1997).

Alaskan BMI communities appear to adhere to the glacial models detailed previously in Section 2.2. In the headwaters of the glacier-fed Atigun River, Alaska, for example, sites close to glacier margins were overwhelmingly dominated by seven species of *Diamesa* (Tilley 1978). In contrast, Andrews and Minshall (1979, in Oswood 1989) note a range of just 5.3-23.8-percent Diptera across 11 Rocky Mountain alpine streams; a temperate pattern borne out by numerous other studies (e.g. Minshall 1981; Death and Winterbourn 1995; Delong and Brusven 1998). Plecoptera and Ephemeroptera are the next most abundant in the benthic fauna of Alaskan river systems; Trichoptera are generally scarce. Also notable are several orders that are either absent or at very low abundance in Alaskan rivers and streams: Hemiptera, Odonata, Megaloptera, Neuroptera, semi-aquatic Coleoptera, and representatives of several families of Plecoptera (Pteronarcyidae, Peltoperlidae and Perlidae), genuinely lotic Coleoptera (e.g. Psephenidae and Elmidae), the Trichoptera family Hydropyschidae, and the Ephemeroptera family Ephemeridae (Oswood 1989).

The recently deglaciated Wolf Point Creek offers an interesting look at colonization within an Alaskan cold-water stream in Glacier Bay National Park, Alaska (Flory and Milner 2000). *Diamesa* were the first taxa to colonize and subsequently showed a strong negative correlation in their densities with water temperature – intolerance of temperatures greater than 4°C may be responsible for their summer declines in abundance at this site (Flory and Milner 2000). Taxa other than *Diamesa* only appeared as the relative influence of non-glacial waters increased and sediment load and turbidity declined (Flory and Milner 1999; Flory and Milner 2000; Milner et al. 2001).

Dorava and Milner (1999) sampled the macroinvertebrate communities of the Johnson River, a short 26 km glacier-fed river on the west side of Cook Inlet. The community was characterized by low densities, averaging 460 inidividuals per m^2 , and low diversity with only three main taxa, Chironomidae,dominated by the subfamily Diamesinae, the mayfly *Baetis*, and the stonefly *Plumiperla*. As a contrast, Dorava and Milner (2000) collected macroinvertebrate samples downstream from Skilak Lake on the glacier-fed Kenai River in southeastern Alaska, on the other side of Cook Inlet, with the major difference being the presence of large lakes. Lower Kenai River macroinvertebrate densities typically exceeded 2000 per m^2, with a diversity of 12 taxa, which included four Plecoptera taxa (*Pluimperla* sp., *Isoperla* sp., *Paraleuctra* sp. and *Capnia* sp.), four mayfly taxa (*Baetis* sp., *Drunella* sp., *Ephemerella* sp. and *Cinygmula* sp.) and a number of caddisflies (*Hydropsyche* sp. and *Glossoma* sp.). Dorava and Milner (2000) explain that the presence of the two lakes improves the downstream river conditions for salmon and benthic macroinvertebrates by intercepting suspended sediments, thus supplying outflows that are less turbid. Glacial systems with lakes also supplement downstream winter low-flows and dampen or attenuate peak flows (Dorava and Milner 2000). Additional sampling at two sites downstream of the upper lake on the Kenai River, Kenai Lake, in 1998 showed moderate macroinvertebrate densities (approximately 1000 and 1300 individuals/m^2) with higher proportions of Chironomidae (88-percent), mostly comprised of Diamesinae and Orthoclaadinae (Dorava and Ness 1999). In terms of the glacial river conceptual model (Milner et al. 2001), Kenai River benthic communities reflect the change seen with increasing distance from the glacial source, with chironomid dominance further upstream, and a more diverse community of stoneflies, mayflies, and some caddisflies downstream.

Sampling on the glacier-fed Talkeetna River at a station 5 miles upstream from the confluence with the Susitna River, revealed an estimated density of nearly 800 individuals/m^2 (Frenzel and Dorava 1999). An estimated 23 taxa of invertebrates were collected, with dipteran flies (mostly chironomids) accounting for 78 percent of the collection, with 32 percent of the sample comprised of the chironomid genus *Orthocladius* sp. Seven EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa were identified in the qualitative sample. This composition seems to suggest an invertebrate community as seen in the upper positions of the conceptual model by Milner et al. (2001).

Although there remains a dearth of studies, the diversity of taxa in the hyporheic zone of Alaskan glacial streams appears both diverse and distinct from fauna typically encountered on the benthos (Wesener et al. 2011). In a study of a glacial stream in the Tongass National Forest, Wesener et al. (2011) found that copepods and oligochaetes were dominant taxa in the hyporheic zone; these taxa are thought to indicate high rates of surface water infiltration (Dole-Olivier et al. 1994). In studies of European glacial rivers, copepods were exclusively found in the hyporheic zone (Malard et al. 2003); however, Wesener et al. (2011) identified copepods in benthic samples as well suggesting there may be substantial connectivity between surface waters and hyporheic zones for similar streams.

The literature on periphytic communities in Alaskan glacial river systems is limited, but general principles and patterns in primary productivity have emerged from a Canadian Rockies study. In southwestern Canadian glacial streams, Geseirich and Rott (2012) found that diatoms were well represented in all types of glacier streams even with peak flows in summer with very variable runoff, and very close to the glacier mouth. The dominant diatom species were a pioneer taxa *Achnanthidium minutissimum* and the fast-water species *Hannaea arcus*. In the east and southeastern Alps, environmental conditions were highly variable, but more than 60-percent of the most common diatoms from a large sample of glacial streams (Rott et al. 2006) were similar to those found by (Gesierich and Rott 2012). These findings supported the hypothesis that glacially influenced streams contain a specific group of diatom and algal taxa.

2.4. Susitna River studies (1980s)

In the 1980s, a series of reports on the physical and biological attributes of the Susitna River were developed in conjunction with the then-proposed development of an earlier Susitna Hydroelectric Project [Susitna-Hydroelectric (Su-Hydro) Project (FERC No. 7114)]. Historic studies were conducted by the Alaska Department of Fish and Game (ADF&G), which included evaluations of the BMI community. (Friese 1975; Riis 1975; Riis 1977; Schmidt et al. 1983; Hansen and Richards 1985; Van Nieuwenhuyse 1985a, 1985c; Trihey and Associates 1986). The following is a review of those 1980's studies that addressed benthic invertebrate and algal fauna in the Susitna River.

2.4.1. Benthic Macroinvertebrates

Studies conducted in the 1980s focused on benthic macroinvertebrate communities in the sloughs, side channels, and tributaries of the Middle Segment of the Susitna River from river mile (RM) 125 to RM 142 during the open-water period from May through October. Efforts included direct benthic sampling with a Hess bottom sampler and drift sampling. Alaska Department of Fish and Game (ADF&G) efforts in 1982 and 1984 also involved collection of juvenile salmon in these side channels and sloughs, and an analysis was conducted to compare gut contents with the drift and benthic sampling results (Schmidt et al. 1983; Hansen and Richards 1985). In addition, Hansen and Richards (1985) collected water velocity, depth, and substrate-type data to develop habitat suitability criteria (HSC), which were used to estimate weighted usable areas for different invertebrate community guilds, groupings based on behavioral type (swimmers, burrowers, clingers), in slough and side channel habitats. Efforts in 1985 (Trihey and Associates 1986) expanded to include sampling at nine sites in the Middle Susitna River Segment: three side channels, two sloughs, two tributaries, and two mainstem sites.

Schmidt et al. (1983) collected invertebrates from kick screen and drift net sampling in 1982, in two sloughs (Sloughs 8A and 11) and two tributaries (4th of July Creek and Indian River). Sampling results were reported as part of electivity index analyses in conjunction with the study of juvenile salmon food habits, but not otherwise summarized in the report. Percent frequencies of the different invertebrate taxonomic groups in the drift and kick samples from each date and site, with results focused primarily on drift sampling. Drift density or drift rate estimates from kick net samples were not presented, nor were density estimates from kick screen samples; consequently, quantitative conclusions are difficult to make without a re-analysis of the data. Nevertheless, a qualitative review of the tabular data suggests that chironomid larvae, pupae, and adults were the dominant items collected in drift, often representing in combination well over 50 percent of the items, but the relative proportions varied considerably among the collections. Some exceptions were noted. For example at Slough 11 on September 5, 1982, capniid stonefly nymphs represented 90 percent of the invertebrates collected in drift net samples. Statistical comparisons indicated that the proportions of invertebrate taxa collected concurrently by kick screen and drift nets were significantly different. Drift samples contained fewer Chironomidae larvae, and more adult dipterans and other terrestrials than the kick screen samples.

Hansen and Richards (1985) collected benthic invertebrates in the middle Susitna River, as part of an evaluation of available fish food resources and the gain and loss of benthic invertebrate habitat resulting from changes in flow. From May through September 1984, drift and Hess samples were collected at the head and middle sections from four off-channel habitat sites (3 side channels, 1 side slough). The invertebrate community in drift and benthic samples collected during 1984 were relatively diverse with 52 taxa from 14 aquatic or semi-aquatic orders represented as well as eleven non-insect or non-aquatic orders (Hansen and Richards 1985). Chironomids (dipteran midge flies) were common throughout the sampling period and were numerically dominant in both drift and benthic collections. Mayflies were common in benthic samples early in the summer, comprised mostly of Baetidae, Ephemerellidae, and Heptageniidae. Stoneflies from the families Chloroperlidae and Perlodidae were more common in early summer, whereas individuals of Capniidae were more prevalent later in the summer. However, benthic Hess samples collected extremely low numbers of invertebrates; results were presented as total numbers for all collected samples, ranging from 118 individuals from 32 samples (an average of 3.7 individuals/ft^2) to 977 individuals per 20 samples (an average of 48.9 individuals/ft^2) (Hansen and Richards 1985). In general, numbers were higher in the August/September samples than the June/July samples, which is in general agreement with benthic communities in glacial-fed systems. The diversity seen in the samples from this study suggests a benthic invertebrate community as seen in the upper positions of the conceptual model by Milner et al. (2001).

Consistent with the literature on glacial-fed systems, the most frequently occurring invertebrate group in drift samples was the dipteran flies, with mayflies being second most frequent, and stoneflies) being the third (Hansen and Richards 1985). Chironomid flies and baetid mayflies made up the majority of individuals in Diptera and Ephemeroptera, respectively, whereas no family was dominant in Plecoptera. Chironomids were relatively abundant throughout the entire sampling period while mayflies were relatively common only in early June. Hansen and Richards (1985) concluded the composition of invertebrates within drift samples was affected by hydraulic conditions at the head of the side slough or side channel. When mainstem discharge was sufficiently high to breach slide slough channel heads, the amount of drift was higher than under unbreached conditions. The authors suggested that mainstem discharge slightly above the critical breaching flow was more important than higher flow levels because mainstem waters would provide greater amounts of drift without the turbidity to substantially reduce visibility.

Hansen and Richards (1985) also developed depth, velocity, and substrate habitat suitability indices for benthic invertebrate behavioral guilds and used instream flow hydraulic models developed for fish species to develop Weighted Useable Area (WUA) versus flow relationships for the guilds. Depth was not limiting to any of the guilds. Velocities from 0 to 3.0 feet per second (fps) were considered optimal for sprawlers, low velocities were optimal for burrowers, and relatively high velocities (up to 2.2 fps) were considered optimal for swimmers and clingers (Figure 2.4-1). Burrowers were more commonly found in sand and silt substrate, sprawlers and swimmers were commonly found in small gravel to rubble-sized substrate, and rubble substrate was optimal for clingers (Figure 2.4-2). Relationships for WUA versus flow were developed for each of the guilds at four sites: Slough 9, Side Channel 10, Upper Side Channel 11 and Upper Side Channel 21 (Hansen and Richards 1985). In general, WUA increased with increasing flows (e.g., Slough 9). Hansen and Richards (1985) concluded that WUA increases rapidly above the critical breaching flow level, which is the point where mainstem discharge becomes the main controlling factor.

Trihey and Associates (1986) produced a close out report for invertebrate analysis of benthic and drift samples taken in August and September, 1985 at nine sites in the middle Susitna River, from RM 128 to RM 140.2. Their report contained the initial identification and bench sheet tallies, presenting combined counts for all samples at each site, but provided no density estimates (benthic or drift). Chironomid larvae are also missing in the benthic samples, as it is indicated that they were still being identified. These data sheets did reveal that a number of mayfly, stonefly, and caddisfly genera were present in the middle Susitna River, and that different genera were collected in tributaries, as compared to mainstem associated sites.

Baseline field data for benthic primary and secondary production was also collected in 1985, as part of the Primary Production Monitoring Effort (Van Nieuwenhuyse 1985c). Chlorophyll-a (*chl-a*), and macroinvertebrates were collected from early April to late October 1985 from a variety of off-channel and mainstem habitat sites. Early April sampling took place in an open-water lead in Slough 8A, and revealed high macroinvertebrate densities (average 17,600 individuals/m^2) comprised almost entirely of chironomid larvae. Sampling in early May in Slough 8A revealed macroinvertebrate densities averaging 2,950 individuals/m^2, again almost entirely chironomids. Average densities in Slough 8A in August 1985 remained similar to spring levels, with a surge in September 1985 (13,964 individuals/m^2); again, chironomids represented over 80 percent of the numbers.

Results from five mainstem habitat sites, with locations spanning from PRM 105.2 (Whiskers West) to PRM 142.1, showed similar macroinvertebrate numbers, with densities ranging from 393 to 8,820 individuals/m^2 in May 1985, but diversity was considerably greater; chironomids accounted for an average of 53 percent of the density, and only 8 percent of the macroinvertebrate biomass (Van Nieuwenhuyse 1985c). Sampling in August and September 1985 showed low average densities at mainstem sites (44 – 164 individuals/m^2), with large increases occurring in October 1985 (1,729 – 7,109 individuals/m^2).

2.4.2. Periphyton

Algal communities were periodically sampled and analyzed for chlorophyll-a at Susitna Station from 1978 to 1980. In the 1980s, algae samples were collected as part of the APA Susitna Hydroelectric Project water quality studies, with sampling conducted at Denali, Cantwell (Vee Canyon), Gold Creek, Sunshine, and Susitna Station on the Susitna River, as well as on the Chulitna and Talkeetna rivers (Harza-Ebasco 1985 as cited in AEA 2011). Analysis showed low productivity (less than 1.25 mg/m^3 chlorophyll-a) and indicated algal abundance was most likely limited by high concentrations of turbidity (AEA 2011).

Van Nieuwenhuyse (1985c) also collected chlorophyll-a from substrate surfaces from early April to late October in a variety of off-channel and mainstem habitat sites as part of the Primary Production Monitoring Effort. Early April sampling occurred in an open-water lead in Slough 8A in pool, riffle, and run habitat. Chlorophyll-a densities in the pool ranged from 4.0 - 132.4 mg/m^2, averaging 36.4 mg/m^2. Riffle and run habitats supported similar epilithic algal growth as in the pools, but displayed greater diversity, with riffles dominated with filamentous algae (*Hydrurus foetidis*), and chlorophyll-a ranging from 4.6 – 132 mg/m^2 (Van Nieuwenhuyse 1985c). Runs in Slough 8A supported *Zynema* and *Spirogyra* along the margins, and *Hydrurus* and a variety of pennate diatoms in higher velocities, with chlorophyll-a densities ranging from 5.1 – 81 mg/m^2. Returning to Slough 8A in early May, a riffle and run section, which was ice-covered three weeks earlier, recorded chl-a densities of 78 mg/m^2 in the run, and 4.8 mg.m^2 in the riffle (Van Nieuwenhuyse 1985c).Additional sampling in mainstem sites dominated by pre-breakup flows in May 1985 showed lower chl-a densities. A side channel in Slough 9A site (PRM 131.5) averaged 1.02 mg/m^2, mainstem margins at PRM 142.1 averaged 1.31 mg/m^2, and mainstem channels at PRM 120.6 and PRM 117.4 averaged 1.47 mg/m^2. Sampling at PRM 105.2, termed Whiskers West, took place in a side channel that existed, pre-breakup, as a 100-m long open-water lead pool, with warm groundwater, and a dense layer of periphytic growth, with chlorophyll-a density averaging 20.3 mg/m^2. Algae samples beyond May 1985 were not analyzed; therefore, no data were available for summer or fall.

3. CONTROLLING FACTORS FOR BENTHIC COMMUNITIES

As summarized in Section 2, the abundance and community structure of benthic organisms in large river systems and in glacial-fed river systems are determined by numerous environmental factors. It is important to note from these primary concepts that the key factor influencing the riverine communities is flow. Within large river systems, flow is a major determinant in how resources, such as nutrients and organic matter, are distributed, which affects the benthic communities. Within a glacial-fed river system, seasonal flows influenced by glacial melt strongly affect water temperature, turbidity, and substrate stability within the river system, which in turn dictates the benthic community structure and distribution.

These factors are often interrelated; for example, flow in combination with local geology largely determines the substrate composition within the river (Armitage 1984), and substrate characteristics set many of the basic conditions for habitat space, food, and protection for benthic invertebrates Minshall 1984). The source (glacial, groundwater, overland) of flow components, their degree of mixing, the seasonality of floods and ice processes, turbidity and shading by riparian vegetation all contribute to patterns of water temperature which in turn affect benthic invertebrate distribution and their growth, vital rates and development patterns (Armitage 1984; Wallace and Anderson 1996; Ward 1992; Williams and Feltmate 1992). Large rivers characteristically form networks of surface and groundwater flow paths that are both laterally and longitudinally extensive. The energy of flowing water constantly reshapes the form of these intertwined pathways (Ward 1994; Ward and Stanford 1995; Poff et al. 1997) creating a shifting habitat mosaic of channel and floodplain structures which are the essential architecture supporting riverine food webs. The distribution and composition of riparian vegetation arises through the dynamic interplay of flow, substrate and the life history characteristics of the vegetation itself and in turn can alter local flow conditions, reshape channels through the contrasting processes of sediment accumulation and erosion and shape benthic communities through both these physical effects and inputs that serve as food. Flood pulses are thought to be one of the primary environmental factors influencing the ecology of large rivers, providing cues that initiate biological events (e.g. drift, migration, spawning) and consequently drive ecological processes. The dominant influence of flow on all aspects of river ecology is intertwined through each of the following components: temperature, substrate, light and turbidity, riparian vegetation, organic matter, and nutrients.

3.1. Flow

The natural flow regime of rivers, which includes the magnitude, timing, frequency, duration and variability of floods (Figure 3.1-1.; Poff et al 1997), largely controls channel morphology and biological productivity (Bayley 1995; Stanford et al. 1996). The flow regime is a key driver of the ecology of any riverine system and not only creates and maintains habitat structure through interactions with local geology, but also the associated physical processes that can be expressed in biologically meaningful ways. Bunn and Arthington (2002) stated it well, “Flow is a major determinate of physical habitat in streams, which in turn is a major determinant of biotic composition.”

Variation in flow plays perhaps the dominant role in regulating the distribution, abundance and coexistence of BMI across all types of lotic systems (Hoffman et al. 2006). Hart and Finelli

(1999) conducted a review of the "multiple causal pathways" by which flow affects benthic organisms. The five principle pathways these authors identified are dispersal, predator-prey interactions, competitions, resource acquisition, and habitat use (Figure 3.1-2).

Flow as a controlling mechanism of drift is, perhaps, the most important contributor to colonization of new or disturbed substrates (Williams and Hynes 1976, 1977; Townsend and Hildrew 1976; Minshall and Petersen 1985; Hart and Finelli 1999). Upon settlement from drift dispersal, benthic organisms will encounter the substrate with a variety of local habitat characteristics. These habitat characteristics also are affected by flow, in terms of the erosional and depositional processes that determine bed form and composition, as well as channel sinuosity and pool-riffle sequences (Hart and Finelli 1999) and set the stage for biotic interactions (competition, predation, resource partitioning) that also influence community composition and abundance.

Variability in flow, from floods or spates to drought, is the major source of natural disturbance in lotic waters and can influence macroinvertebrate abundance and taxa richness (Hershey and Lamberti 2001; Hart and Finelli 1999). In high flow disturbances, macroinvertebrate communities can be scoured by suspended sediments, crushed by bed load, or simply swept away downstream by irresistible velocities. In contrast, low or declining flows, such as droughts, lead to reduced habitat availability for macroinvertebrates, thus potentially intensifying biotic interactions while fragmenting longitudinal and lateral ecological processes that depend on water movement for transport of resources and biota (Resh et al. 1988; Lake 2000).

Benthic macroinvertebrate communities generally recover rapidly from natural disturbance, implying the existence of spatial refugia. These refugia may either be in the hyporheic areas of the substrate, (Williams and Hynes 1976; Lake 2000; Fowler and Scarsbrook 2002; Gray and Fisher 1981), or in portions of the streambed known as hydrodynamic dead zones, such as behind larger or stable substrates, in which bed shear stress shows little to no increase during high flow disturbances (Lancaster and Hildrew 1993; Matthaei et al. 2000). It is important to note that flow changes are not always considered detrimental. Flow disturbance can play a central role in determining the structure of stream communities (Resh et al. 1988; Poff 1992; Lake 2000). The occurrence of one or more natural high flow disturbances in river systems is generally considered a normal forcing factor that maintains species diversity and influences the carrying capacity of fish and BMI communities. As addressed in Section 2, even extreme flood events fit into the natural flow paradigm (Poff et al. 1997), serving to re-shape existing channel structure, add habitat diversity and complexity, and maintain floodplain ecology. This suggests that periodic, stochastic large-scale disruption of aquatic systems is necessary to maintain the long-term viability of certain taxa or communities (Sparks et al. 1998). Wootton et al. (1996) for example demonstrated via a multitrophic model that removing flood disturbance from a river would increase predator resistant grazing insects that would divert energy away from food-chain linkages to predatory fish.

The intermediate-disturbance hypothesis (Connell 1978) predicts that communities will exhibit higher taxa diversity when subjected to intermediate levels of disturbance, as opposed to those communities undergoing either greater or lesser perturbation. Originally proposed as a mechanism explaining diversity patterns for communities in tropical rain forests and coral reefs, Ward and Stanford (1983b) adapted this principle to predict and describe diversity patterns observed in natural and altered lotic ecosystems. They maintained that in "undisturbed" lotic systems, fluctuations in discharge and water temperature sustain a non-equilibrium community, with different taxa thriving as environmental conditions change (Ward and Stanford 1983b). Low disturbance environments that display a relative constancy of stream temperature and streamflow such as that exhibited by spring-fed streams, or from dams with hypolimnetic releases, may result in lower taxa richness and diversity through intensified biotic interactions and the expression of competitive dominance (Ward and Stanford 1979, 1983b). Instances of systems with high disturbance regime might be a stream below a hydroelectric dam that exhibits large, daily flow fluctuations (e.g., facilities operated on a "peaking" basis – see Section 4.1) and in such cases also may result in lower taxa diversity.

3.2. Temperature

Water temperature in river habitats varies over a broad range of spatial and temporal scales (Langen et al. 2001; Liu et al. 2005; Thorp et al. 2006), and influences aquatic biological communities and ecosystem processes (Vannote et al. 1980; Pringle et al. 1988; Power et al. 1988; Milner and Petts 1994; Kocik and Ferreri 1998; Thorp et al. 1998; Milner et al. 2001; Malard et al. 2002). Temperature influences rates of physical and biological processes, and in part, controls distribution, growth, life history, and survival of fish (Lantz 1970; Power et al. 1988; Eliason et al. 2011), and benthic invertebrates (e.g., Newell and Wayne 1978). Water temperature is also a strong determinant of primary production (Raven and Geider 1988; Salmaso and Zignin 2010).

Factors that control water temperature depend on scale and geography (Fausch et al. 2002; Gu and Li 2002, Garner et al. 2013), but also on the nature of anthropogenic alterations in the environment (Caissie 2006; Olden and Naiman 2010). Climate, geology, shading, channel form, groundwater exchange, inflow from tributaries, longitudinal position, and air temperature influence water temperature regimes in rivers (Sullivan et al. 1990). For example, water temperature in a glacier-fed river basin in New Zealand was determined to be influenced by water source contribution (precipitation or melt), hydroclimatological conditions (air temperature, precipitation, and stream discharge), lineal distance from the source, stream flow volume, channel morphology, and riparian vegetation (Cadbury et al. 2008). Land use activity, such as agriculture, can influence water temperatures (Young et al. 2005), and flow alterations from dams in particular are known to strongly influence the thermal regime in rivers (Bunn and Arthington 2002). Thus, to understand how water temperature influences patterns in biological processes, auxiliary physical, biological, and anthropogenic factors in the landscape that influence the water temperature regime must be well-understood (Fausch et al. 2002; Hamilton et al. 2012).

In turn, physical factors such as water chemistry, streamflow regime, riparian vegetation, land use, lithology, and source can all interact to influence water temperature and biological communities among the mosaic of habitats in river ecosystems (Petts 1986; Stanford and Ward 1993; Polis et al. 1997; Welch et al. 1998; Nakano et al. 1999; Nakano and Murakami 2001). Each trophic level in the riverine food web is affected, either directly or indirectly, by water temperature, from the decomposition of organic matter (Dang et al. 2009), to the bioenergetics of invertebrates (Vannote and Sweeney 1980) and juvenile salmonids (Beauchamp 2009). In freshwater, benthic macroinvertebrates (BMI) are the primary prey source of aquatic species at upper trophic levels, such as juvenile salmonids, but are also primary consumers of lower-level organisms, such as algae. Metamorphosis timing (Hynes 1976; Gregory et al. 2000; Harper and Peckarsky 2006) and production rates (Newell and Wayne 1978) of BMI are reliant on water temperature regime, and shifts in abundance and timing of BMI have direct effects on interactions among trophic levels, altering food web dynamics among species (e.g., Paine 1980).

At the base of the food web, water temperature directly affects production and distribution of algal organisms in riverine environments. Morin et al. (1999) found that periphyton production was strongly related to temperature in streams, and in a laboratory experiment, DeNicola (1996) showed that different species of benthic algae thrived at different temperatures. Water temperature correlates with seasonal variables, such as photoperiod, nutrient availability, and flow, and plays a role in seasonal peaks and troughs in phytoplankton abundance in river habitats (Sullivan et al. 2001). However, the magnitude of the influence of water temperature on algal growth depends on the cumulative interactions of water temperature and other physical factors. For example, water temperature, flow, dissolved matter, and turbidity in a spring-fed system in Oregon were constant, so changes to the algal community could not be attributed to those variables, and were likely more influenced by light intensity and photoperiod (Sherman and Phinney 1971). In more extreme environments, temporal stability in these variables might lead to increases in production of algae. For example, increased standing crop of benthic algae in a subarctic stream in Alaska was attributed to the comparatively stable nature of physical (temperature and flow) and chemical characteristics (high alkalinity and phosphorous concentration) in that stream (LaPierre et al. 1989). Finally, temperature may have less impact on growth of algae when light and nutrients are the limiting factors, but may have more of an effect on growth where these factors are not limiting (Raven and Geider 1988).

For BMI, water temperature affects all aspects of development and growth including the duration of embryonic (Elliot 1972) and larval development (Ross and Merritt 1978; Vannote and Sweeny 1980), larval activity levels (Hynes 1970), and emergence timing (Hynes 1976; Gregory et al. 2000; Harper and Peckarsky 2006). In general, the duration of embryonic and larval development for BMI decreases as temperature increases, and with some variation, the optimal temperature for development is near the upper lethal limit for each species (Resh and Rosenberg 1984). However, most experimental studies designed to evaluate the effects of temperature on development of BMI have been conducted using constant temperatures. In river systems, thermal experience of BMI fluctuates constantly over time with streamflow and other factors, such as hyporheic exchange (e.g., Fowler and Scarsbrook 2002). Accordingly, Resh and Rosenberg (1984) noted that use of a mean temperature to represent fine-scale temporal fluctuations in water temperature may not lead to biologically accurate results in terms of time- and temperature-dependent development in BMI (e.g., Taylor 1982).

As poikilotherms, growth of BMI is also affected by water temperature. In the Henry's Fork catchment in Idaho, growth of a stonefly *Pteronarcys californica* was greater in a warmer location than in a cooler location (Gregory et al. 2000). The effect of thermal regime on growth of BMI varies among species. In the Flathead River in Montana, Perry et al. (1987) found that populations of two different species of stoneflies responded differently to thermal conditions altered by flow regulation. Altered flow conditions led to larger nymph size, and affected growth and emergence timing in one of the species, whereas the other species responded by having lower population densities and larger nymph size, but growth and emergence timing were not affected. The timing of thermal experience also influences growth in BMI. Numerous studies in unregulated systems have shown that adult mayflies emerging during spring are significantly larger than those that emerge during summer (e.g., Vannote and Sweeney 1980; Wise 1980). Thermal regime also has an effect on growth-dependent reproductive success in some BMI species. For example, in the order Odonata, smaller individuals in a population have adapted to seasonal shifts in the thermal regime to grow faster prior to emergence so they can attain maturity and engage in synchronous reproductive behavior (Procter 1973). However, this phenomenon has not been observed in all species that exhibit synchronous reproduction (e.g., Sweeney and Vannote 1981).

Rivers fed by glacial meltwater function differently than snowmelt or groundwater fed systems (Steffan 1971). Milner and Petts (1994) based their classification of glacier-fed systems on variations in temperature and the degree of influence of ice-melt, snow-melt and rain-dominated flows. Glacial river systems are characterized by low annual water temperatures (< 10°C) (Rothlisberger and Lang 1987; Smith et al. 2001). Water temperature is determined by the proportional contributions of glacial melt (Tockner et al. 1997; Malard et al. 2006), and increases with distance from glacial inputs, as tributaries, groundwater, and rainfall contribute to flow (Smith et al. 2001).

In contrast to rivers in temperate regions, species diversity of BMI communities in glacial waters is low (Milner and Petts 1994; Brittain and Milner 2001; Snook and Milner 2001), and this is due, in part, to temperature-dependent limitations on colonization rates (Flory and Milner 2000). As described in Section 2, Milner and Petts (1994) presented a conceptual model of the longitudinal distribution of BMI taxa in glacier-fed rivers that is structured by gradients in temperature with distance from glacial margins. Milner et al. (2001) determined that taxa in glacial rivers displayed a threshold temperature below which they did not occur, and significant breaks occurred for various groups at 2, 4 and 8°C. The lower temperature threshold for Orthocladiinae, Tipulidae and Oligochaeta was 2°C, whereas Perlodidae, Taeniopterygidae, Baetidae, Simuliidae, and Empididae had a lower threshold of 4°C, and Nemouridae, Leuctridae, Heptageniidae, Rhyacophilidae, and Chironominae had a lower threshold of 8°C.

3.3. Substrates

In aquatic habitats, substrate includes everything on the bottom and sides of the aquatic environment as well as everything projecting into aquatic habitat (Voshell and Wright 2002). Substrate is the base on which an organism lives, providing habitat space, food, and protection (as flow refugia); it is the “stage upon which the drama of aquatic insect ecology is acted out” (Minshall 1984, p. 358). Substrate acts both directly on insects as the medium supporting their benthic life history and indirectly as a major modifier of their environment. Substrate can be composed of inorganic mineral material ranging from large boulders to fine clay particle, as well as many kinds of organic material, including fallen trees, filamentous algae, detritus, and living rooted plants. Sedentary insects require attachment sites for successful feeding, growth and pupation. One of the earliest patterns noted by aquatic ecologists was that the total abundance of benthic animals varied with substrate type and that most taxa showed affiliations with particular substrate characteristics (Andrews and Minshall 1979; Minshall 1984; Cobb et al. 1992; Bourassa and Morin 1995; Schröder et al. 2013), often dictated by particular physiology and behavioral adaptations (Jowett et al. 1991; Collier and Halliday 2000; Holomuzki and Biggs

2000). Case building organisms need materials from which to build their shelters, burrowing organisms are restricted to substrates they can penetrate, vascular plants provide the best substrate for clinging invertebrates, while other organisms require loose particles with interstitial spaces they can penetrate as refuges from both flow and predators (Rabeni and Minshall 1977; Minshall 1984).

Isolating the role substrate plays in determining benthic community structure or composition is complicated by interactions with multiple other environmental variables. Current velocity, particle size, dissolved oxygen content, and the size and amount of organic matter all tend to co-vary, which makes the attribution of causality from association in field studies difficult (Death 2003; Fuller et al. 2011; Jones et al. 2012). Flow interacts strongly with substrate types to determine what habitat is available to macroinvertebrate communities in a stream or river. Where water is slow moving or still, substrate may be relatively homogenous; where currents are active or flow is swift and turbulent, more diverse and heterogenous substrates will occur (Minshall 1984; Allan and Castillo 2007). Fine particles can blanket the surface of the streambed at low flows, effectively restricting algal growth and impeding the movement, feeding actions and respiration of aquatic insects. Fine sediments may also fill in spaces between larger particles, limiting access by insects as well as the movement of water and dissolved gasses through the substrate. The structure of river bottoms is thus inextricably linked to variations in flow.

There is a common and marked association between biomass of primary producers and invertebrate consumers and the degree to which the substrate is free of fine sediment (Osmundson et al. 2002; Jones et al. 2012). Without the large flood pulses of most natural water regimes, species with life stages that are sensitive to sedimentation, such as the eggs and larvae of many invertebrates, can suffer high mortality (Poff et al. 1997). Potential impacts of sedimentation on invertebrate communities include alteration of substrate composition causing changes to the suitability of the substrate for some taxa (Bourassa and Morin 1995; Collier et al. 1998); increases in drift due to sediment deposition or substrate instability (Henley et al. 2000; Gibbins et al. 2010a; Gibbins et al. 2010b); and reduced respiration due to the deposition of silt on respiration structures (e.g. gills) and impairment of filter feeding (Jones et al. 2012).

Flow also interacts with substrates in terms of channel stability. Stability, in the context of substrate, refers to the degree of resistance to movement. The stability of substrate is the result of the interplay between flow, existing channel morphology and substrate types. In general terms, freshwater ecologists have noted that diversity and abundance of BMI tend to increase with substrate stability (Cobb et al. 1992; Holomuzki and Biggs 2000; Castella et al. 2001). Negative correlations between increasing discharge and substrate particle movement can result in density declines on the order of 94-percent (Cobb et al. 1992).

For glacial systems, Milner et al. (2001) found that the channel bottom measures, rated as part of the Pfankuch Stability Index (PSI; Pfankuch 1975), along with water temperature explained the most variation in their model predicting BMI taxa richness and density in glacial rivers. On a scale of 15 to 60, with higher values being more unstable, substrates with PSI channel bottom scores greater than 40 were dominated by Diamesinae, Orthocladiinae, Simuliidae, and Oligochaeta. Six taxa (Leuctridae, Perlodidae, Rhyacophilidae, Chironominae, Empididae, and

Limoniidae) had less than 5-percent occurrence with PSI channel bottom scores greater than 40 (Milner et al. 2001).

Because smaller particles can be overturned more frequently, stability is generally proportionate to particle size, which may explain the general finding that benthic organisms increase in numbers and diversity over a sequence of increasing particle size (Cummins and Lauff 1969; Rabeni and Minshall 1977; Minshall 1984; Matthaei et al. 2000; Barnes et al. 2013; Schröder et al. 2013). This general relationship is also seen for algae, an important food resource for BMI, which can further explain the increased abundances on larger substrates. Substrate structure and stability can influence algal communities, with filamentous algae and taxa that grow in large mats found with larger substrates, and smaller, motile taxa such as diatoms found on finer sediments (Burkholder 1996; Cattaneo et al. 1997). Periphyton and biofilms are usually more productive on stable substrates, avoiding scouring and general abrasion from substrates during higher flow events (Stevenson 1990; Biggs and Smith 2002; Allan and Castillo 2007).

Bed sediment movement is one of the primary factors limiting the biomass of periphyton in gravel-bed rivers or rivers with flashy discharge (Biggs and Close 1989; Biggs et al. 1999; Francoeur and Biggs 2006b). Flow-substrate interactions further constrain algal growth in glacial rivers through light limitation from high turbidity and from a low supply of growth-limiting nutrients (Lloyd et al. 1987). High flow periods can thus result in severe limitation of benthic algae in main channel and upstream connected channel habitats fed by glacial meltwaters through the interaction of bed instability with reduced light availability and low temperatures (Malard et al. 2006). Rempel et al. (2000) attribute the limited algal and grazer productivity and low representation of grazers in the zoobenthos of the Fraser River, Canada to high turbidity and fine sediment deposition.

While larger substrates generally result in increased BMI and algal abundance and diversity due to their relative stability, coarser bed materials also provide more interstitial spaces for macroinvertebrates to use as refugia, as well as for the trapping of detritus for food (Rabeni and Minshall 1977; Minshall 1984; Cobb et al. 1992; Bourassa and Morin 1995). The shape, size, and distribution of substrate can influence the type, composition and quantity of allochthonous organic matter that is retained and available as food (Hoover et al. 2010) or create difficult to access pockets where biofilms can escape herbivory (Bergey and Weaver 2004), while the decomposition of large accumulations of organic particles can create anoxic zones. Substrate type and arrangement thus has the potential to set both the physical habitat conditions for benthic organisms as well as creating the trophic conditions under which biotic interactions occur (Minshall 1984; Allan and Castillo 2007).

The influence of substrates on benthic communities extends right down to the surface of the substrates themselves, where texture and availability of crevices influences individual particle's suitablility as habitat. Several studies have investigated the relationship of colonization and substrate texture, and found greater diversity and abundance on irregular surfaces than on smooth substrates (Hart and Finelli 1999; Downes et al. 2000; Boyero 2003; Bergey and Weaver 2004; Bergey 2005). Surface roughness and crevices can protect benthic invertebrates from high-discharge events, abrasion, desiccation, and predators (Downes et al. 2000; Boyero 2003; Allan and Castillo 2007), and can influence algal colonization, algal size distribution, and species composition (Schneck et al. 2011). Increased substrate texture can result in increased algal biomass by protecting algae from grazers and disturbances (Dudley and D'Antonio 1991; Bergey 2005).

3.4. Turbidity and Suspended Sediments

Turbidity is a measure of water clarity, describing the degree to which light traveling through a water column is scattered by suspended organic and inorganic particles in a water column (APHA 1999). The scattering of light increases with a greater suspended load of materials, which may include soil particles (clay, silt, and sand), algae, plankton, microbes, and other substances (Bilotta and Brazier 2008).

The effects of inorganic turbidity and suspended sediments on aquatic habitats and the biota depend upon the concentration, the duration, the chemical composition, and the particle-size distribution of suspended particles (Henley et al. 2000; Bilotta and Brazier 2008). High and sustained levels of sediment may permanently alter benthic community structure, diversity, density, biomass, growth, and rates of reproduction and mortality. Many of these changes are expressed via the food webs and habitat, with the influence of increased sediment in the system beginning at the primary trophic levels (algae, macrophytes, phytoplankton) passed on directly to primary consumers, typically benthic macroinvertebrates (Henley et al. 2000; Bilotta and Brazier 2008).

Turbidity most affects the benthic community by limiting the amount of light penetrating through the water column, thus restricting the rate at which periphyton and emergent and submerged macrophytes can assimilate energy through photosynthesis (Lloyd 1985; Lloyd et al. 1987; Henley et al. 2000; Bilotta and Brazier 2008). Van Nieuwenhuyse (1983) quantified the negative relationship between turbidity and light penetration for streams in the Birch Creek and Chatanika River drainages of interior Alaska that were subject to placer mining activities. He found that extinction of light was directly related ($r^2 = 0.99$) to increased mining-induced turbidity, and also used this extinction coefficient to calculate (+5-percent) the percentage of incident photosynthetically active radiation (PAR) that penetrated to a particular depth. PAR is the portion of the light spectrum (400 to 700 nanometers) that is useful to plants for photosynthesis. Van Nieuwenhuyse (1983) estimated that the amount of light reaching a depth of 0.1 m ranged between 75 and 79-percent of that available at the surface at turbidities of 0.50-10 NTUs, from 60 to 69-percent at turbidities of 25-50 NTUs, and from 0.3 to 5-percent at turbidities of 500-1,000 NTUs.). Van Nieuwenhuyse (1983) concluded that, for interior Alaskan streams, the demonstrated importance of light in controlling streams productivity points to increased turbidity as the single most important disruption to that productivity.

Van Nieuwenhuyse and LaPerriere (1986) also described a strong positive relationship ($r^2 = 0.67$) between gross primary productivity and light penetration in shallow streams (up to 0.5 m in depth). Calculations estimated that the primary productivity of shallow, clear-water streams could be reduced by about 3 – 13-percent with a turbidity of only 5 NTUs. An increase of 25 NTUs in shallow streams would decrease primary production by 13 - 50-percent. In clear streams with depths greater than 0.5 m, the primary production would be reduced even further (Van Nieuwenhuyse and LaPerriere 1986). At high concentrations and durations with higher velocities (e.g., floods, spates), suspended sediment also has the potential to physically scour periphyton away from streambed substrates, as well as being abrasive and damaging to the photosynthetic structures of macrophytes (Alabaster and Lloyd 1982; Steinman and McIntire 1990;(Uehlinger et al. 1996; Francoeur and Biggs 2006a) (see also Sections 3.1 Flow, and 3.3 Substrates).

The effects of turbidity on benthic invertebrates are less well-defined. Most studies examining suspended sediments have focused simply on the effects of the deposition of those fine sediments on benthic invertebrates, or do not separate out the effects of the sediments in suspension from the sedimentation effects (Ward 1992). Generally, high levels of suspended sediments may exert a number of direct and indirect effects on aquatic organisms (Ward 1992). These potential effects include abrasion or clogging of respiratory structures, reduced feeding rates or efficiency, reduced vision, exposure to toxins adsorbed by the particles, reduced vision, and induction of drift (Ward 1992). Invertebrates that graze periphyton for their energy and nutritional requirements will likewise by affected by any changes in suspended sediment concentrations that adversely affect algal growth, biomass, or species (Newcombe and MacDonald 1991).

Many studies have shown that increased suspended sediments typically increase invertebrate drift in river systems, reducing both abundances and diversity within the affected reach (Rosenberg and Weins 1978; Gray and Ward 1982; Wagener and LaPerriere 1985; Culp et al 1986; Doeg and Milledge 1991; Shaw and Richardson 2001; Bilotta and Brazier 2008). Gammon (1970) investigated a small, low-gradient stream receiving suspended sediments from a quarry, and found that suspended sediment levels of less than 40 mg/L above normal levels, while resulting in no significant deposition, resulted in a 25-percent reduction in macroinvertebrate abundance in benthic samples, most likely due to emigration by drift. Suspended sediment levels of 120 mg/L or more above background levels reduced densities by 60-percent. In addition, Gammon (1970) found shifts in benthic invertebrate communities at suspended sediment concentrations as low as approximately 53 mg/L, largely characterized by increases in silt-tolerant genera, such as the mayfly genus *Tricorythodes*. Culp et al (1986) investigated the movement of finer sand sediments (0.5 – 2.0 mm in diameter size) that could directly disturb the benthic macroinvertebrate community through abrasion, as opposed to the habitat alteration and indirect impact caused by deposited sediments. Sands were added to the riffles along a third-order coastal stream with enough tractive force to transport the sand particles along the bottom of the stream, rather than allowing for deposition. This abrasive sediment transport caused catastrophic drift, occurring either immediately or shortly after the sediment addition, resulting in losses in benthic invertebrate density of 50-percent and greater in a 24-hour period, and a large change in the community composition (Culp et al. 1986). This demonstrates that even when flows are insufficient to suspend fine sediments, transport of sediments along the bottom and across the substrates still has the potential to act as a community-level disturbance at lesser flows.

In Alaska, glacial meltwater naturally contributes enormous amounts of fine suspended sediment and consequent turbidity to major drainages (Lloyd et al. 1987). Glacial rivers typically have turbidities in excess of 30 NTU, due to the high concentrations of fine sediment and glacial flour (Milner and Petts 1994). Suspended sediments in glacial rivers range in size from 0.001mm to 1 mm, however sediment smaller than 0.063 mm (glacial flour), derived from glacier-grinding, dominates most glacial rivers (Milner and Petts 1994). High turbidity may limit instream primary production in glacial rivers, especially in the middle zone (orders 4 – 6) of the fluvial network where autotrophy might be expected to dominate (Vannote et al. 1980).

Turbidity in a glacial system is tied to the melting of the glacier for supply and thus is highly seasonal with a distinct and predictable flood pulse (Tockner et al. 2000). Between early summer and autumn, the increased radiation and temperature-controlled release of glacial melt increases the bed load transport, high concentration of suspended solids (high turbidity), and relatively low temperatures (Milner & Petts 1994). During the early summer melt period in the Alps of Europe, peak suspended sediment concentrations of over 500 mg/L are typical, and concentrations above 2000 mg/L are common (Hieber et al. 2001). In winter, while stream channels are clear of the glacial fine sediments, channels often are covered by snow, subject to freezing, or are dewatered (Malard et al. 2006). Uehlinger et al. (2010) suggest that light exclusion by snow may be an important factor responsible for low periphyton biomass during winter. Snow cover during winter is presumably more efficient in intercepting light than turbidity; measurements in the European Alps showed that a snow-pack of 60 cm may reduce PAR light by 99-percent (Rott et al. 2006; Uehlinger et al. 2010).

In contrast to the harsh environment in summer and winter, more benign conditions prevail during spring and autumn when flow is relatively low (no bed load transport), temperature moderate, and light conditions are little affected by the suspended sediments from glacial melting. Studies suggest that the spring and autumn periods are ecological “windows of opportunity” for benthic algae, in that these are periods favoring the greater algal biomass otherwise constrained by factors such as moving bed sediments and limited light availability (Uehlinger et al. 2002; Milner et al. 2009). Uehlinger et al. (2010) quantified algal biomass in five glacial and three non-glacial streams in the Swiss Alps over an annual cycle. Biomass peaked during the autumn/early-winter period, between late September and January. In spring, however, the favorable conditions of low flow, low turbidity, and a lack of coarse sediment transport were not paralleled by an increase in periphyton biomass. Uehlinger et al. (2010) suggest that increased UV radiation, in combination with low temperature and relatively high PAR intensities, and/or phosphorus limitation may suppress the periphyton accumulation in spring despite otherwise suitable flow conditions. Contrary to those findings, Rott et al. (2006) reported spring periphyton peaks in a glacial stream of the Austrian Alps, but noted that in glacial streams, biomass/chlorophyll peaks are short in timing, to fit the limited window of physical restrictions. In the non-glacial streams investigated, biomass was significantly higher than levels in glacial streams, although seasonal periphyton patterns were similar (Uehlinger et al. 2010). In two of the glacial streams studied, no snow cover existed, resulting in relatively high biomass amounts persisting until March/April, suggesting that with increased PAR levels and non-turbid waters, the winter period can be productive for algal growth.

The assessment of the peaks in algal production in the Swiss Alps by Uehlinger et al. (2010) agrees with the initial assessment made by Van Nieuwenhuyse (1985b) regarding the Susitna River. Based on initial sampling efforts in 1984, Van Nieuwenhuyse (1985a) stated that the major period of organic carbon production in the middle Susitna River occurred during the fall period (September and October), when mainstem flows are generally lower (typically below 12,000 cfs, rarely greater than 22,000 cfs), and turbidity levels declined to less than 50 NTU. Van Nieuwenhuyse (1985b) also stated in an impact analysis that during this fall period mainstem and peripheral channels along the entire length of the middle Susitna River display a large pulse of primary production, which he estimated would exceed the spring pulse by at least two orders of magnitude on a daily system-wide basis. This algal production during the previous fall transition period supports benthic macroinvertebrate production throughout the winter, resulting in the large numbers of emerging aquatic insect (especially chironomid and plecopteran) adults observed in mid March to mid-May 1985 throughout the middle river, indicating that substantial secondary production takes place during the winter months (Van Nieuwenhuyse 1985b).

3.5. Role of riparian vegetation in aquatic ecosystem function

River ecosystems are intimately connected to and interact with their surrounding watershed (Hynes 1975; Naiman and Decamps 1997) and this connection is now generally acknowledged to be reciprocal (Baxter et al. 2005; Marcarelli et al. 2011). Flow regimes alter both channel and vegetation dynamics while riparian vegetation reciprocally creates and stabilizes fluvial habitats (Corenblit et al. 2007; Hannah et al. 2007), filters hyporheic water and influences runoff patterns (Tabacchi et al. 1998), is the major source of material for aquatic organic matter cycles (Zah and Uehlinger 2001; Eggert and Wallace 2003; Acuña et al. 2004; Solomon et al. 2008; Kiffney and Richardson 2010; Langhans et al. 2013) and provides habitat, migration paths, food and shade for both terrestrial and aquatic organisms (Naiman and Decamps 1997; Davies et al. 2013) including macroinvertebrates (Aguiar et al. 2002). At the same time, the river provides water, sediments, and nutrients and acts as a vector for seed transportation to potential colonization sites on the riverbanks (Camporeale et al. 2013). Above-ground biomass of vegetation modifies flow-fields and can retain sediment while below-ground biomass alters both hydraulic and mechanical properties of the substrate and thus affects erosive potential (Gurnell 2013).

In a general sense, the condition of upland vegetation, and the structure of the riparian corridor in particular, play a fundamental role in mediating and controlling these patterns; vegetation dynamics are the major factor maintaining linkages between river and floodplain as well as between the riparian corridor and the uplands (Tabacchi et al. 1998). These linkages are dynamic and subject to frequent change as a consequence of geomorphic, hydrological and anthropogenic events (Marcarelli et al. 2011; Cross et al. 2013). A major driver of these linkages is the seasonal or episodic pulsing of river discharge (see Section 2.1) which determines both the degree of connectivity and the rates of exchange processes across river-floodplain gradients. Successional processes in the riparian corridor and floodplain are reset by hydrological disturbance which varies in intensity and frequency both longitudinally from upstream and downstream but also laterally across floodplains. Although compositional shifts across the riverscape (sensu Junk et al. 1989) are thought to control ecosystem processes and biodiversity at multiple temporal and spatial scales (Benke et al. 2000; Tockner et al. 2000; Malard et al. 2006), many aspects of these relationships are still poorly understood.

3.5.1. Riparian vegetation as a source of energy and matter

Most river systems are substantially heterotrophic for much of their extent (Wallace et al. 1995; Zah and Uehlinger 2001; Lennon and Pfaff 2005; Solomon et al. 2008; Solomon et al. 2011), relying on inputs of nonliving organic material (detritus) from the riparian zone and overall catchment for the energy that drives the aquatic foodweb (Tank et al. 2010). Numerous studies have demonstrated that most of the particulate organic matter and a large component of the dissolved organic matter is derived from areas adjacent to the stream channel (Perry and Perry 1991; Kiffney et al. 2000; Lennon and Pfaff 2005; Parkyn et al. 2005; Yonekura et al. 2009; Berggren et al. 2010; Richardson et al. 2010; Watkins et al. 2010). When organic matter originates outside the system in this way, it is termed allochthonous (e.g. leaves falling from streamside vegetation into a river) whereas organic detritus that originates from within the system is termed autochthonous (e.g. aquatic macrophytes or algae). Inputs of terrestrial organic matter are an important source of carbon for microbial and macroinvertebrate communities that use leaf litter directly as either food or substrate as well as for higher trophic levels such as macroinvertebrate predators and fish.

In small streams, allochthonous inputs from riparian vegetation typically dominate energy budgets (Kiffney et al. 2000; Acuña et al. 2004; Kiffney and Richardson 2010; Romito et al. 2010; Marcarelli et al. 2011) because of their narrow channels and overhanging vegetation, but even large river food webs can be driven by substantial quantities of terrestrial origin organic matter. Changes in the geometry of larger river channels (decreased perimeter to area ratios) dictate that the quantities of allochthonous materials that fall directly into the river channel are a smaller fraction of the total organic matter relative to the potential for instream production or transport of material from upstream reaches (Thorp and Delong 1994; van der Nat et al. 2002; Doering et al. 2007). Indeed in large rivers the majority of terrestrial inputs may be derived from upstream reaches (Caraco and Cole 2004; Wipfli and Musslewhite 2004; Wipfli 2005; Wipfli and Baxter 2010) or from periodic inundation of associated floodplains (Hein et al. 2003; Langhans et al. 2013) rather than through direct vertical inputs from bankside vegetation.

Substantial numbers of both correlative studies and experimental manipulations have demonstrated that invertebrate biomass and production are tightly linked to both the quantity (Wallace et al. 1997; Wallace et al. 1999; Eggert and Wallace 2003; Johnson et al. 2003) and quality (LeRoy et al. 2006; Kominoski et al. 2011; Bruder et al. 2011; Grattan and Suberkropp 2011; Marcarelli et al. 2011) of allochthonous inputs. Many invertebrate taxa show life history phenologies and growth patterns which track the abundance of allochthonous inputs (Yeates and Barmuta 1999; Albarino and Balseiro 2002; Wipfli and Musslewhite 2004; Kochi and Kagaya 2005; Going and Dudley 2008; Leberfinger et al. 2010; Richardson et al. 2010). Riparian plant diversity also appears to have impacts on both macroinvertebrate (Snyder et al. 2002; Leroy and Marks 2006; Lecerf et al. 2007b; Schindler and Gessner 2009) and microbial (Dang et al. 2005; Fernandes et al. 2013) community composition and structure with associated shifts in organic matter processing rates in streams. Quality variation in detritus from *within* riparian plant species whether due to hybridization and genetic variation (LeRoy et al. 2006; Axelsson et al. 2010), variable growth conditions (Lecerf and Chauvet 2008) or herbivory (Irons et al. 1991; Molvar et al. 1993), has also been shown to affect the interactions of stream biota with allochthonous litter inputs. Terrestrial material may ultimately contribute less to the gross generation of secondary biomass than autocthonous production because it is mostly stable, and resistant to being broken down (recalcitrant), whereas instream primary production is typically more bio-available (labile) if less abundant (Thorp and Delong 2002).

3.5.2. Channel and bank stability, instream habitat

Many streams and rivers receive a steady input of woody branches and tree trunks that originates almost entirely from the riparian zone (Collier and Halliday 2000; Pusey and Arthington 2003; Helmus and Sass 2008). Over time these large woody inputs can form debris dams which interact with variability in hydrology to increase the complexity of channel morphology and provide useful habitat for stream biota (Rabeni and Minshall 1977; Minshall 1984; Gurnell et al.

2005; Helmus and Sass 2008). These interactions can be enhanced by additional factors such as the frequency of flooding events or the presence of beavers (Anderson and Rosemond 2010; Jones 2012) or other ecosystem engineers. Passive ecosystem engineering by vegetation has frequently been identified as the driver of fluvial habitat morphogenesis. A common example of this is the development of characteristic vegetated islands (Gurnell and Petts 2006; Corenblit et al. 2007; Bertoldi et al. 2010; Welber et al. 2012). Uprooted trees and other large woody debris form depositional areas for sediments, eventually leading to fixing of substrate and the evolution of larger established islands (Gurnell et al. 2005; Gurnell and Petts 2006). Islands form within the constraints of flood dynamics, and rates of both sedimentation and the growth capacity of woody vegetation; they develop most quickly where woody taxa (such as widespread pioneering species of *Populus* and *Salix*) have the ability to sprout from vegetative fragments (Corenblit et al. 2007). The permanence and distribution of these islands is dictated by the occurrence of large infrequent floods, the cycle of which establishes a dynamic biotic-abiotic feedback between the physical disturbance regime and substrate cohesion and surface roughness controlled by biotic elements (Corenblit et al. 2007).

Riparian forests can also be seen as providing ecosystem services to aquatic communities in the form of temperature regulation and bank stabilization. Overhanging vegetation reduces solar heating of stream water (Beschta 1997; Tabacchi et al. 1998) in low order streams although thermal regulation of stream temperatures is almost certainly dominated by other factors in larger rivers (Larson and Larson 1996) (see Section 3.2 Temperature). The role of riparian vegetation in fixing channel and bank stability has also been extensively studied. Riparian vegetation alters flow fields, fluvial transport processes, and the morphology of the river through mechanisms including: (1) increases in riparian bank strength due to plant roots (Smith 1976; Abernethy and Rutherfurd 2000), (2) the influence of vegetation on flow turbulence (Pollen et al. 2004), and (3) the general effects of vegetation on soil water balance and the combined effects of these processes on riparian erosion and sedimentation (Camporeale et al. 2013). Vegetation can clearly also facilitate erosive effects in fluvial environments through mass failure and fluvial scour. The impact of vegetation on bank stability processes, both stabilizing and erosive, combine to affect overall river migration rates and the formation of meanders and braiding patterns (Abernethy and Rutherfurd 2000; Pollen et al. 2004; Corenblit et al. 2011; Camporeale et al. 2013), which structures the macro-scale distribution of habitats for aquatic taxa.

In Wolf Point Creek in Glacier Bay National Park, Alaska, the establishment of deciduous riparian vegetation through time (and with distance from the source of glacial melt) allowed additional groups, such as the Trichoptera, to establish (Flory and Milner 1999). Leaf litter inputs from riparian vegetation are a necessary food source for many obligate or facultative shredding organisms (Richardson 1991; Jonsson et al. 2001; Kochi and Yanai 2006). Taxa commonly associated with willow catkins and leaves (including *Brillia, Dicranota* and Limnephilid caddis flies) colonized following the development of willows along the riparian corridor (Flory and Milner 1999; Flory and Milner 2000). The trophic complexity of streams typically increases with accumulation of allocthonous materials as riparian vegetation develops, linking invertebrate community composition and abundance to the dynamics of vegetation succession and establishment (Gíslason et al. 2001; Wipfli 2005; Kiffney and Richardson 2010).

3.6. Organic Matter and Nutrients

Organic matter containing carbon (C) and organic and inorganic forms of major nutrients, nitrogen (N) and phosphorus (P), are the essential building blocks of life in river ecosystems. Carbon is essential to meet the energy needs of heterotrophic organisms and N and P are components of amino acids, DNA, RNA and phospholipids essential for cellular processes. N and P supply can enhance the rate of primary production in streams and rivers while the quality and supply of C, N, and P from organic sources is important for regulating secondary production, the growth rate of higher trophic levels including invertebrates and fish.

Rivers are essential links between terrestrial and marine ecosystems that receive, process and transport carbon and nutrients from land to the sea. Organic matter in particulate (POM) and dissolved forms (DOM) is the energy source of food webs in river ecosystems and originates from a variety of sources. Terrestrial or allochthonous inputs of plant litter, including woody debris and leaf litter, are often the dominant C source in headwater streams where shading limits algal production. As dictated by the “River Continuum Concept” (Vannote et al. 1980; see Section 2.1), as streams become larger, carbon sources from algal production can become more important as shading is reduced, and the quality of carbon can also change along river corridors as plant litter is colonized, degraded, and consumed by microbes, fungi and invertebrates.

Since carbon and nutrients are essential to primary and secondary production, it is also important to understand how C, N and P cycles are linked. Organic matter such as plant litter that falls into the river is decomposed by microbes and fungi, invertebrates directly consume plant material as well as microbial and algal biofilms, and fish can directly consume plant material, microbial and algal biofilms or invertebrates (Figure 3.6-1.). Organic N and P is regenerated back into inorganic forms through mineralization, while fish and invertebrates excrete reduced phosphorus and ammonium. These inorganic forms of N and P are directly available for algal production. Carbon and nutrients cycle through living and non-living aquatic compartments, including organisms, river sediments and as dissolved and particulate material in the water column. This fundamental principle in stream ecology called “nutrient spiraling” describes cycling in the context of downstream transport where cycles are stretched into spirals (Newbold et al. 1981). This concept was later expanded to include carbon (Newbold et al. 1982).

In addition to downstream transport of fluvial C, N and P, marine-derived carbon and nutrients from the decay of returning adult salmon have been shown to enrich both aquatic (Bilby et al. 1996, Cederholm et al. 1999) and terrestrial (Drake et al. 2006, Hocking and Reimchen 2006) and act as a positive feedback to fish populations (Naiman et al. 2002). Many examples of marine-derived carbon and nutrient delivery have been shown in Alaskan anadromous streams and lakes (Wipfli et al. 1998, Chaloner et al. 2002, O’Keefe and Edwards 2003, Hicks et al. 2005, Schindler et al. 2005, Lessard and Merritt 2006, Hood et al. 2007, Fellman et al. 2008).

Both the quantity and quality of terrestrial inputs interact with other factors to determine the abundance and composition of benthic invertebrates. Aquatic consumers will react differently to changing terrestrial plant composition and this has the potential to alter critical rates of ecosystem functioning such as decomposition as well as modify energy transfer rates to higher trophic levels (Garcia et al. 2012). There is a strong positive correlation between the abundance of shredding invertebrates and subsequent processing rates for terrestrial organic matter from river systems around the world (Wallace et al. 1997; Whiles and Wallace 1997; Hutchens and Wallace 2002).

Clarifying the relationships between invertebrate abundance and decomposition rate is further complicated because litter of different tree species varies in chemical composition and thus its perceived quality as a food resource by benthic invertebrates (Haapala et al. 2001; Kobayashi and Kagaya 2004; Leroy and Marks 2006). The sequential shift from early colonizers to upland vegetation types across large river floodplains attracts and supports different communities of aquatic invertebrates and microbial decomposers (van der Nat et al. 2002; Lecerf et al. 2005; Paetzold et al. 2006; Kominoski et al. 2011) with subsequent differences in decomposition rates of litter species alone or in complex mixtures (Schindler and Gessner 2009; Bruder et al. 2011).

The vast majority of studies of terrestrial litter inputs into aquatic systems have focused on the contributions of deciduous shrub and tree leaves (Wipfli 2005; Tank et al. 2010). In open canopied systems dominated by riparian wetlands (such as beaver complexes), grasses may represent a substantial fraction of terrestrial inputs to streams (Shaftel et al. 2011; Kostka 2012). Largely thought to be a poor food source for macroinvetebrates because of their high C:N ratios, grasses may nonetheless play a substantial role in stream metabolism where they dominate the riparian communities of stream ecosystems . Decomposition dynamics of this material may depend on interactions with other riparian species, for example Shaftel et al. (2011) documented low rates of decomposition of bluejoint grass (*Calamagrostis* Canadensis) litter in streams of the Kenai Peninsula that increased with stream N concentrations associated with the percentage of *Alnus spp.* cover in the watershed.

In glacial rivers, snowmelt and glacial meltwater are dominant sources of flow, as well as organic matter and nutrient fluxes. Studies of glacial rivers have shown that water sources including groundwaters, snowmelt and glacial water contribute to the expansion of the hydrological network with complex changes in the spatial and temporal pattern of organic matter and nutrients (Tockner et al. 2000; Zah and Uelinger 2001; Malard et al. 1999). For instance, meltwater was the primary source of particulate organic carbon (POC) and inorganic sediments in the main channel of the Val Roseg River in the Swiss Alps, but hillslope groundwater was an important source of dissolved organic carbon (DOC) to side channels and sloughs (Tockner et al. 2002). Consistent with the river continuum concept (see Figure 2.1-1), organic matter in glacial rivers also changes longitudinally from the glacial terminus to the downstream floodplain ecosystems. In small proglacial streams, particulate organic matter input increased from glacial terminus to downstream reaches and changed from direct litter inputs to lateral POM inputs in floodplain (Zah and Uehlinger 2001).

Glacial rivers are often considered to be harsh environments where high-turbidity and low quality organic matter sources can limit river production (Hodson et al. 2005). However, recent studies have shown a diverse ecology of glacial streams related to variation in carbon and nutrient sources and the physical environment of main channel and off-channel habitats. Side sloughs fed from groundwater sources with low turbidity can maintain much warmer temperatures through the summer months when disconnected from main channel flows (Uehlinger et al. 2003). Conversely, when off-channel habitats are hydrologically connected during high flows, they may receive organic matter sources from the main channel as well as floodplain sources (Tockner et al. 2000). This interplay between inundation at high flow and contraction at lower flows, known as the flood pulse concept (see Section 2.1), may be particularly important in glacial ecosystems where with a wide variety in habitats suitable for primary and secondary production.

While glacial weathering has long been recognized as a source of inorganic carbon and phosphorus (Tranter et al. 2002, Tockner et al. 2002), recent studies have shown that glaciers may also be a source of bioavailable organic matter. Hood and Berner (2009) demonstrated that glaciers in Southeast Alaska were a source of old and bio-available (labile) organic matter to streams in Southeast Alaska. Nevertheless, at the landscape level DOC and inorganic N concentrations are negatively related to glacial coverage across a range of Southeast Alaska streams (Hood and Berner 2009), indicating that terrestrial plants and soils are perhaps more important sources for the total flux to the coastal zone. Consequently, large glacial rivers may contain a variety of organic matter with varying lability from both old and new sources.

4. HYDROPOWER FLOW OPERATIONS

A regulated stream is one in which the natural flow regime has been purposely altered or controlled, generally via construction of a dam, to meet an anthropogenic purpose, whether it be for flood control, water supply, or hydroelectric generation. The effects of these types of projects on downstream ecosystems can range widely depending on operational flow regimes, and geographic and climatologic setting. This section will focus on a specific class of operational effects that are most often associated with hydroelectric operations, pulse-type flows (PTF). The effect of pulse-type flows are related to the sharp and sudden increase in flows (e.g., pulse) for a relatively short period of time and then a decrease back to the original flow. These types of flow patterns may be related to power operations as well as to meet specific resource objectives, such as for fish and recreational purposes. The sections that follow will describe the more common PTF operations used in hydropower.

4.1. Types of Pulse-Type Flows

Pulse-type flows (PTF) can occur in response to power generation needs as well as to meet specific resource objectives such as provision of recreation flows, flushing flows, attraction flows, and others. Given that the patterns of flow releases below projects can differ dramatically, the resulting effects on downstream ecosystems will differ as well. In this section, the most common types of PTF that are associated with hydroelectric projects are described. In this discussion, the term “baseflow” is used to refer to the flow that occurs just prior to and after the PTF cycle rather than the low flow condition that typically represents the groundwater contribution to a river system.

4.1.1. Power Peaking Flows

Hydroelectric projects that operate as peaking facilities are designed to meet increased demands for power during certain periods of the day and reduced demands at other times. Peaking operations typically result in daily cycles of increasing flows during morning hours to some level sufficient to meet demand, sustained flow at that level for a certain period of time, followed by a reduction in flow as demand goes down. This power peaking pattern often only occurs during the weekdays; reduced power demand on weekends relegates operations to more of a baseload condition in which flows remain steady (Figure 4.1-1). The overall magnitude of flow change between the baseflow and peak flow can be quite large and can result in stage differences on the order of feet for some projects. These short-term flow fluctuations result in the repeated dewatering and re-inundation of those shoreline areas and the fluctuating current velocities over submerged substrates, creating a “freshwater intertidal zone,” also called the varial zone (Figure 4.1-2), which is characterized by reduced invertebrate density and diversity, as well as low algal production (see Section 5.1.4.1. for a detailed discussion). As flows ramp up and down, there are generally few regulations on the rate of flow change during the up-ramp cycle (exceptions generally related to safety considerations associated with angling, recreation, etc.), but the rate of flow reduction (down-ramp) is often specified as part of project operations.

4.1.2. Load-Following Flows

Another type of PTF that is related to increased power demands are flows associated with "Load-following." Load-following can result in real-time changes in flow releases to match real-time shifts in power demand; in essence, flows are regulated to match increasing or decreasing power loads that can occur throughout a 24 hour period. Oftentimes, load-following is integrated directly into a peaking operation, but they can be separate depending on FERC license conditions for a given facility. Load-following can result in large fluctuations in flow over relatively short time intervals. Multiple cycles can occur within a day, for example, as seen in historical operations of the Kerr Dam on the Flathead River, Montana (Figure 4.1-1), or load-following effects of the Baker Project on the lower Skagit River, Washington (Figure 4.1-3). Load following effects depicted as sharp increases-decreases in flow are evident at other times in Figure 4.1-1 and are associated with the daily peaking cycle.

In general, the same categories of impacts as noted above for peaking flows can occur with load-following operations; i.e., varial zone formation delineated by the upper and lower flow cycle (Figure 4.1-2). Given the frequency and magnitude of fluctuations, impacts may actually be greater under load-following than under straight peaking operations. Similar to peaking, load-following operations are typically not restricted in how fast flows can be increased, but are often regulated as to how quickly they can be reduced.

4.1.3. Flushing Flows and Channel Maintenance Flows

Another category of PTF is the programmed release of flows designed to mobilize and transport sediments from stream segments below a dam, often called "flushing flows," or to maintain channel form and function, termed "channel maintenance flows." Both can result in a rapid increase in flows up to a predetermined level where they are maintained for a specified period of time (determined sufficient to achieve sediment transport objectives – typically 1-7 days), and then are reduced to baseflow conditions. The magnitude, duration and frequency of these types of PTF are highly dependent on resource management objectives, ambient sediment conditions, and project specific operations (Reiser et al. 1989; Kondolf and Wilcock 1996).

Graphically, a flushing flow is similar in pattern to a peaking cycle that would remain at the high flow for several days before decreasing (Figure 4.1-4). The frequency of these kinds of PTF is much less than peaking flows; once per year or less is generally sufficient unless a catastrophic input of sediment occurs requiring additional prescriptive flows. The release of a flushing flow or channel maintenance flow is typically timed to be synchronous with normal runoff processes so its effects are ecologically compatible and beneficial to the existing aquatic biota. However, different release timings may be needed to offset catastrophic sediment influx.

Depending on the management objectives, the magnitude of these types of flows can range from large (sufficient to mobilize the bed and flush sediments at depth) to moderate (sufficient to mobilize surficial fine sediments). However, the duration and frequency of these types of PTF are generally short (1-7 days, 1 time per year); any effects to biota can likely be reduced if the rate of flow reduction (downramping) is low. The short-term cycle of a programmed flushing flow or channel maintenance flow release does not allow the formation of a defined varial zone.

4.1.4. Recreation Pulse Flows

Flows for recreation-based activities such as rafting, kayaking, boating, fishing, etc. represent another kind of PTF. These types of flows can range widely in their magnitude, frequency, and duration, depending on project layout and operational constraints. For some projects, recreation flows may be tightly scheduled and confined to certain times and even days of the year, for others they may be integrated directly into hydroelectric operations such as peaking or load-following, or they may be scheduled on an almost ad hoc/opportunistic basis. An example of a project where recreation flows are fully integrated into project operations is the Nantahala Gorge on the Nantahala River in North Carolina, where a robust economy actually relies on the daily peaking operations of an upstream hydroelectric project to provide whitewater recreation opportunities, on an almost year round basis. For that system, recreation flows are provided during the day in conjunction with the release of flows for power peaking, while flows are reduced during the evening when power demands are lower. The Clackamas River Project in Oregon represents a system where recreation flows for one of its regulated tributaries (Oak Grove Fork) are being considered on an ad hoc basis (as high flow conditions may allow), to provide for kayaking, even though such conditions may not occur on an annual basis. Other projects specify recreation flows on an annual basis during certain times of the year. For these types of projects, the recreation flow perhaps best resembles a series of short duration pulse flows similar to a flushing flow, but that are scheduled over a two – three month period (Figure 4.1-4).

4.1.5. Outmigration Flows

The release of a block of water during the spring months to support the outmigration of anadromous salmonid smolts and fry represents another form of PTF that is practiced in the Pacific Northwest and California Coastal areas. Perhaps the best example of this occurs in the Columbia River Basin of the Pacific Northwest where for many years there has been a systematic and coordinated release of flows (April – June) from dams throughout the basin as a means to facilitate the outmigration of smolts through the series of mainstem dams (CPMPNAS 1996). The release pattern for these flows exhibits a sharp increase (generally via spill) up to a certain level of flow, sustaining that flow for several weeks to months (depending on flow availability) and then decreasing the flows down to the baseflow (non-spill) condition.

4.1.6. Adult Attraction Flows

Under some conditions, hydroelectric and other water projects may provide a short duration flow release to stimulate and promote the upstream migration of adult anadromous fish. These generally target fall spawning fish whose migration patterns can occur coincident with natural low flow conditions and elevated water temperatures. Since adult movements are often stimulated by a rapid increase in flow (spate), a series of short duration (1-7 days depending on water availability) pulse releases of flow can be useful in stimulating upstream movements as well as providing some thermal benefits. In some cases, pulse flows of a sustained nature have been recommended. For example, to promote adult migration in the lower Klamath River, Zedonis et al. (2003) proposed three options of PTF, including one sustained pulse flow lasting four weeks (designed to provide thermal benefits), a series of short duration (1-2 day) pulse flows, and a hybrid of the two consisting of a series of short-duration pulses during the first part of the month followed by a reduced but sustained pulse release for the remaining period (Figure 4.1-5). The range in flow fluctuation associated with these PTF was from around 450 cfs (baseflow) to 1500 – 2000 cfs, a 3.5 to 4 fold increase. In regulated streams, the release of these types of PTFs generally coincides with programmed flow conditions that are typically the lowest of the year. Hence, the extent of dewatered channel margins is at its greatest and correspondingly with the release of PTF during this period substantial rewatering of the channel occurs.

4.1.7. Thermal Flows

For some hydroelectric projects, PTF are released at certain times of the year or under certain extreme conditions to specifically provide thermal benefits for fish. For example, coldwater releases from Shasta Dam to the Sacramento River are meant to match the thermal requirements of winter run Chinook salmon for spawning and egg incubation. In this case, the releases are on more of a sustained period (throughout the spawning and egg incubation period) rather than a series of pulses. However, one of the license requirements for the Madison River project in Montana requires that when water temperatures reach a certain level, a series of pulse flows are to be released during the cooler late evening hours to provide thermal benefits during the day. In general, depending on the flow release configuration of the dam (e.g., surface-spill; selective gates; hypolimnetic), the thermal characteristics of PTF can vary widely and must be considered relative to effects on BMI and fish communities.

4.1.8. Baseload Adjustments

In general, the flow release patterns from flood and water supply dams can be relatively constant over long periods of time (several months), with changes made only to accommodate system (reservoir filling) needs or to meet specific habitat objectives (e.g., increased flows during salmon spawning periods). The same is true of many hydroelectric projects that are operated as baseload facilities for which power generation is set at some constant level (based on powerhouse capacity) that essentially mirrors natural flow conditions (e.g., run-of-river project) and/or that is consistent with reservoir management objectives (e.g., flood-storage, lake level management for recreation, etc.). However, even these baseload projects that can maintain stable flows for long periods of time, require periodic flow adjustments.

This is illustrated in Figure 4.1-6, which depicts the annual hydrograph of the Kerr Dam hydroelectric project on the Flathead River, Montana, that is operated as a baseload facility. Such adjustments are typically associated with seasonal or monthly adjustments that target aquatic species life history needs such as spawning or rearing, that target reservoir management, or that attempt to mimic some percentage of the natural hydrograph. With respect to the latter, the general pattern of flow change is from a baseflow condition during the late summer through winter, increased flows during the spring, and then tapering back down to baseflow conditions. In some systems, the regulation of flows creates a temporal shift in the flow regime whereby flows during certain times of the year become higher than normal, for example during the drafting of a reservoir to create flood storage, and then become lower than normal during other times of the year such as whenever the reservoir is filling (Figure 4.1-7). Flow adjustments based on aquatic biota life stage needs may take on a slightly different pattern, with increases occurring during both spring and fall months to accommodate spring and fall spawning fish, while flows at other times (winter and summer) may focus on egg incubation and juvenile rearing.

Although technically not a PTF, to the extent the baseload adjustments result in a rewatering or dewatering of channel margins, they can potentially affect BMI and fish communities. with PTFs, it is the reduction in flow that is of most concern to BMI and fish as it can result in loss of productive habitats as well as stranding and trapping of organisms. In the case of baseload adjustments, such reductions are generally relatively small compared with the range of flow fluctuations associated with power peaking and load following. For example, flows recommended to provide for salmonid egg incubation are typically in the range of 2/3 of the flows provided for spawning (Thompson 1974). These types of adjustments would generally not result in large expanses of channel margins becoming dewatered, and therefore the loss to BMI production would likely be minor. However, if the reductions are rapid, some stranding of both invertebrates and fry and juvenile fish could occur. Ramping rate restrictions on the rate of flow reduction serve to reduce the potential for these types of impacts.

5. EFFECTS OF HYDROPOWER ON BENTHIC COMMUNITIES

Many reviews have been written on the effects that hydropower operations have upon on the benthic communities residing below dams (Ward 1976; Ward and Stanford 1979; Armitage 1984; Petts 1984; Cushman 1985; Saltveit et al. 1987; Dewson et al. 2007; Svendsen et al. 2009; Elosegi et al. 2010; Poff and Zimmerman 2010; Tockner et al. 2010). Most of these reviews consider alteration of the flow and temperature regimes among the most important change affecting BMI below dams (Ward and Stanford 1979; Armitage 1984; Bunn and Arthington 2002). However, as this review of the factors controlling benthic communities (Section 3) revealed, the interaction of flow with other factors such as temperature, substrates, water chemistry, riparian vegetation help to define BMI and algae communities in river, both regulated and unregulated.

The following section gives a review of the effects of hydropower flow operations on benthic communities from the standpoint of each of the controlling factors previously discussed in Section 3. The purpose of this section is to show how hydropower operations can affect each of these physical factors (flow, water temperature, turbidity, riparian/organic matter, and nutrients), in turn affecting benthic communities. A conceptual model is first presented to show how flow is altered and how altered flows can affect benthic communities, followed by a description on how regulation can alter water temperature, turbidity, riparian zone/organic matter contributions, and nutrient cycling, and show how each of these affected factors influence the benthic community.

5.1. Flow

In one of the earliest reviews, Ward (1976) recognized four different types of flow regimes that river regulation can produce: a reduced flow, seasonal flow constancy, an increased flow, and short-term flow fluctuations (Figure 5.1-1). Over the course of the year, streams below dams can be subjected to more than one of these altered flow regimes. However, the type of dam and its mode of operation determine the frequency and magnitude of disturbance to the benthic community below the dam.

5.1.1. Seasonal Flow Constancy

Seasonal flow constancy is the result of storing water in a reservoir during the peak runoff periods, and then releasing the stored water during periods of normally lower flows. This regime creates a more stable flow with the elimination of discharge extremes. Constant flows influence physical habitat and water quality, which can in turn produce favorable conditions for the benthic community (Armitage 1984). Many of the studies showing benefits of seasonal flow constancy are centered on the effects of sustained low flows during the spring, when flows would normally be increased and variable (Williams and Winget 1979; Saltveit et al. 1987; Casado et al. 1989). A stable flow results in uniform currents, increases in water clarity due to the deposition of sediments in an upstream reservoir, and increased bed and bank stability, which can lead to increased riparian growth and development of epilithic algae (Figure 5.1-1) (Armitage 1984; Ward and Stanford 1979).

Generally, these changes could be considered favorable to BMI (Ward 1976; Armitage 1984), with increased riparian and aquatic vegetation enhancing the food base and providing additional niches for invertebrates. However, there are also some potential negative effects to the benthic community, especially if the flow constancy is associated with one of the other regulated flow regimes, such as flow reduction or short-term fluctuations (discussed below). For example, the absence of spring flooding may lead to a build-up of allochthonous material along the riverbanks, decreasing the availability of this resource to some species, perhaps placing them at a disadvantage. Those BMI that depend upon the fallen leaves and debris as a food source (i.e., collectors and shredders) would be put at a competitive disadvantage (Saltveit et al. 1987). Under the combination of constant and reduced flows, finer sediments may not be redistributed, especially those from unregulated tributaries downstream from the dam (Ward 1976; Armitage 1984). Such accumulations can result in stream habitat alteration by riparian vegetation encroachment (see Section 5.4) and siltation (Ward and Stanford 1979; Robinson et al. 2004), which would further alter the distribution, composition, and abundance of the BMI community (Armitage 1984). Robinson et al. (2004) noted that the River Spöl was subject to constant residual flows of less than 88 cfs year-round, resulting in sedimentation on the streambed, and allowing the formation of side-slope debris fans into the main channel and the establishment of pools upstream of them. The Spöl was dominated with high densities of gammarids and turbellarians, both taxa associated with streams with predictable and stable flows (Armitage 1976, 1977; Wallace 1990; Ward and Stanford 1995; Vinson 2001).

5.1.2. Reduced Flow

Reduced flow is a broad descriptor for the reduction of the discharge flowing from a dam, as compared to historical, pre-dam flows. The resulting effects are decreased current velocity, depth, and wetted width in the river channel (Figure 5.1-1) (Ward 1976; Armitage 1984; Dewson et al. 2007). Lower current velocities allow finer sediment, such as sand or silt, to settle into coarser substrates, filling interstitial spaces and reducing the area available to BMI to reside (Armitage 1984; Moog 1993). The elimination of high seasonal runoff also prevents the flushing of finer substrates, resulting in a loss of heterogeneity in the bed substrates (Ward 1976; Williams and Winget 1979; Dewson et al 2007). Williams and Winget (1979) found that the decreased flows and losses of habitat heterogeneity below Soldier Dam on the Strawberry River, Utah, resulted in the elimination of several mayfly and stonefly taxa.

The loss of wetted width that may result after prolonged flow reductions decreases available habitat, alters habitat suitability, and reduces habitat diversity, which often results in a reduction in benthic production (Williams and Winget 1979; Armitage 1984; Cazaubon and Giudicelli 1999; Lake 2000; Dewson et al. 2007). Reduced flows also result in changes in BMI community composition and diversity. Some taxa preferring slower water velocities would increase in abundance while others that require swift currents for respiration or food procurement would not remain. In situations where flow reduction is combined with flow constancy, densities below dams are greatly increased compared to unregulated reaches, but taxa richness is reduced, limited to those taxa that favor lower currents and stable conditions (Williams and Winget 1979; Zimmerman and Ward 1984; Ward and Voelz 1988; Voelz and Ward 1991).

In contrast to BMI, algal growth can change as a result of reduced below hydropower dams (Bürgi et al. 2003; Jakob et al. 2003; Uehlinger et al. 2003), switching from a low-biomass diatom assemblage to a high-biomass filamentous green algal mat (Dewson et al. 2007). Uehlinger et al. (2003) noted that prior to implementation of a new flood regime that included experimental floods for the purpose of improving in-stream conditions for biota, the biomass of periphyton on the River Spöl was roughly one order of magnitude higher than in similar streams nearby which had unregulated flows. Algal production can be increased in response to increased water clarity due to sediment trapping in the reservoir (see section 5.3), increased temperatures, higher nutrient concentrations, and reduced current velocity, which corresponds with reduced scour and abrasion of algae (Ward 1976; Williams and Winget 1979; Biggs and Close 1989; Barranguet et al. 2005; Dewson et al. 2007; Verspoor et al. 2010; Slemmons et al. 2013).

5.1.3. Increased flow

Increased flow generally refers to a stable increase in the volume of water flowing from a dam as compared to historical, pre-dam flows. These differ from the sudden and often extreme increases in discharges due to power peaking, which are the short-term flow fluctuations explained in Section 5.1.4. Ward (1976) indicates that such increased flow may be the result of the addition of water diverted from another drainage to augment flows, or from the alteration of flow patterns.

A large increase in flow can lead to higher velocities and scouring of the river bottom, eliminating riparian vegetation and epilithic algae from the substrate surfaces (Figure 5.1-1) (Armitage 1984). Bed-scouring flows can also result in dramatic reductions in macroinvertebrate densities (Holomuzki and Biggs 2000; Robinson et al. 2004; Fuller et al. 2011). High flow conditions can reduce the BMI abundance by increasing the drift rate of certain taxa (Brusven et al. 1974; Armitage 1984; Poff and Ward 1991).

It is also possible that a less severe increase in flow could clear away accumulated fine sediments, leaving a coarser, heterogeneous habitat for BMI colonization (Figure 5.1-1) (Ward and Stanford 1979). However, increased flows in the fall and winter periods could result in the potential loss of accumulated organic matter resources from the interstitial spaces within the substrates, eliminating both spatial and food niches and reducing BMI abundance and diversity (Radford and Hartland-Rowe 1971; Trotzky and Gregory 1974). The regime change from lower flow levels to higher sustained flows can also alter the BMI community composition, eliminating those taxa with preferences for slow currents, and favoring taxa that are more adapted to torrential conditions (Radford and Hartland-Rowe 1971; Ward 1976; Brittain and Salveit 1989).

5.1.4. Short-term Flow Fluctuations (Pulse-type flows)

As shown in Section 3, short-term, regulated flow fluctuations, (e.g., pulse-type flows, or PTF) are a common consequence of hydroelectric power generation, and they can have a pronounced, adverse effect upon the benthic macroinvertebrate community (Trotzky and Gregory 1974; Gersich and Brusven 1981; Armitage 1984; Petts 1984; Cushman 1985; Saltveit et al. 1987; Boon 1988; Moog 1993). Essentially, such fluctuations impose the effects of both reduced and increased flows on a reach of river, inflicting a difficult set of conditions upon benthic fauna residing within the affected area. Therefore, those BMI that remain in a river reach subject to short-term fluctuations must be adapted to withstand the rapid changes in discharge (Munn and Brusven 1991). Those taxa feeding under constant flows or in a narrow range of current velocity would be eliminated or competitively disadvantaged (Ward and Stanford 1979). While the magnitude of regulated flow fluctuations may not be as great as those experienced in unregulated rivers, fluctuations that result from power generation occur over diurnal periods rather than seasonal periods, and this rapid rate of change (up- or down-ramping) can directly affect benthic communities

Flows from Wyman Dam on the Kennebec River, Maine varied from a minimum of 275 cfs to maximum discharges close to 600 cfs, severely reducing densities, taxa richness, and diversity at sites immediately below the dam (Trotzky and Gregory 1974). Casado et al. (1989) found similar results in Spain, with benthic macroinvertebrate density, biomass, diversity and taxa richness reduced on the River Tera below the Cernadilla dam and reservoir for over 8 km due to short-term flow fluctuations. The investigation of the benthic macroinvertebrate community below Dworshak Dam, Idaho revealed a severely altered and simplified community, with high abundance, but low taxa diversity of only 2-8 taxa (Munn and Brusven 1991). Approximately 68-99-percent of the community composition was orthoclad chironomids, known to be tolerant of the rapid changes in discharge from the dam.

Some efforts have been made to reduce the extremes of flow fluctuations and examine the ameliorating effect on benthic communities. The Susquehanna River below Conowingo Dam, Maryland experienced daily flow fluctuations from 106 cfs to over 35,000 cfs (Weisberg et al. 1990). In 1982, the minimum flow requirements were raised from April to September to protect fish and invertebrates. Weisberg et al. (1990) compared macroinvertebrate abundance in 1980 to that in 1982, when minimum flow was increased. Total density was greater in 1982 than in 1980, due to an increase in chironomids and caddisflies. Caddisfly larvae were not common in 1980, increasing from 17.3 to 922.3 individuals/sampling basket in 1982. After minimum flow requirements were discontinued, declines of more than three orders in magnitude were seen in chironomid and caddisfly densities. By establishing a higher minimum flow, benthic macroinvertebrates were provided with extra protection from stresses of exposure and desiccation, and a continuous food supply by the current (Weisberg et al. 1990).

One of the most comprehensively documented studies of invertebrate responses to experimental pulsed flows was undertaken on the Spöl River in Switzerland. In 2000-2002, a series of pulsed flows were released monthly in June, July, and August, exceeding the established reduced flow regime (flows of approximately 88 cfs) by a magnitude of 10 to 30 times higher (Robinson et al. 2003, 2004). Robinson et al. (2004) found that the floods reduced macroinvertebrate densities by 14-percent to 92-percent, averaged across habitat types, with the degree of reduction being related to the magnitude of the flood pulse. Results also showed that bedrock habitats provided greater refuge, generally losing fewer invertebrates as a result of pulsed flows. The pulsed flow releases also altered the existing benthic community, with gammarid amphipods becoming less abundant, particularly in pools and riffles after the pulsed flow events; conversely, dipterans (Chironomidae and Simulidae) became more abundant, as did the proportion of mayflies in all habitats following each flood (Jakob et al. 2003; Robinson et al. 2003, 2004).

5.1.4.1. Benthic Community Responses to Pulse-type Flows

Benthic macroinvertebrates react to flow modifications in a variety of ways. Taxa can respond to change by drifting downstream to more suitable conditions, by following the retreating shoreline as waters recede, or by finding suitable flow refugia. Failure to act in response to flow changes often results in stranding and desiccation on exposed shorelines. Review of the literature regarding the effects of PTF on benthic communities found that potential effects of peaking operations on the benthic communities generally relate to:

1. loss of productive habitat within the band width (often termed the "varial zone") of the littoral stream margins that are periodically exposed during the increase and decrease flow cycle (Brusven et al. 1974; Gislason 1985; Benenati et al 1998; Blinn et al. 1998; Bergey et al. 2010);
2. cycles of increased-decreased drift during pulse flows (Brusven and MacPhee 1976; White and Wade 1980; DeVries et al. 2001) potentially resulting in a reduction in invertebrate standing crop in permanently wetted areas; and,
3. stranding and/or trapping of BMI, as well as fry and juvenile fish, within the varial zone during the flow reduction period (Bauersfeld 1978, Hunter 1992, Brusven et al. 1974).

5.1.4.1.1. Establishment of a Varial Zone

Several studies below dams conducting pulse-type flow releases have reported a presence of a varial zone (Gislason 1985; Perry and Perry 1986; Troelstrup and Hergenrader 1990; Blinn et al. 1995; DeVries et al. 2001; Grzybkowska and Dukowska 2002). The impacts associated with the varial zone are illustrated in Figure 5.1-2, which depicts a hypothetical varial zone in both a longitudinal and cross-sectional format. Varial zones are characterized by reduced invertebrate density and diversity (Fisher and LaVoy 1972; Hauer et al. 1989; Dos Santos et al. 1988). Thevarial zone is often visually distinctive in streams subjected to frequent PTF as an area of the streambed that is lighter colored than adjoining substrates. This color distinction is a result of the lack of algae and periphytic growth (Figure 5.1-3).that is characteristic of permanently wetted substrates (Benenati et al 1998; Blinn et al. 1998; Bergey et al. 2010)

Fisher and LaVoy (1972) examined benthic macroinvertebrate communities along a sand/gravel bar below a hydroelectric dam on the Connecticut River in Massachusetts. Fluctuations of approximately 1 m at the bar completely submerged it during high flows, and exposed much of it during low flows. Four zones were established along a transect running from high (Zone 1, exposed 70-percent of the summer) to low (Zone 4, constantly submerged) water marks. Results indicated reduced diversity, biomass, densities, and taxa richness in Zones 1 and 2, and a composition of mostly non-insect taxa, dominated by nematodes, oligochaetes, and the snail *Amnicola*. Chironomidae were the only insect taxon collected in significant numbers in these frequently exposed zones. Metric values and community compositions of Zone 4 did not differ significantly from Zone 3, which was exposed for 13-percent of the summer, suggesting that the benthic community established at those levels was adapted to brief exposure periods.

Brusven and MacPhee (1976) studied the effects of fluctuating river flows on the macroinvertebrate community in the Clearwater River, Idaho below Dworshak Dam. They determined that the downstream benthic insect community prior to filling of the reservoir was similar to that present after. However, margin areas of the Clearwater River that were influenced by power peaking operations were not colonized readily by stoneflies, mayflies, or caddisflies.

Similar to the findings of Fisher and LaVoy (1972), chironomids were the most common insect encountered in habitats that experienced fluctuations in flow.

Blinn et al. (1995) determined that discharge fluctuations during the summer and winter influenced the benthic community in the Colorado River downstream of Glen Canyon Dam, Arizona. The annual mean biomass of macroinvertebrates in a continuously inundated section of the channel was more than four times the biomass of macroinvertebrates in the proximal varial zone. Blinn et al. (1995) also reported a reduction of over 60 percent in *Cladophora* and epiphytic biomass after five consecutive days of 12-hour exposures in the Glen Canyon Dam tailwaters. Benenati et al. (1998) examined the algal communities within the varial zone and the continuously inundated channel below the baseflow levels. Results showed that *Cladophora* was the dominant algal species on cobbles below the baseflow and *Oscillatoria* was dominant on cobbles in the varial zone. In general, the *Cladophora* biomass decreased under intermittent drying conditions by more than 70 percent, while Oscillatoria declined under permanently wet conditions by up to 50 percent (Benenati et al. 1998). The study also found that diatom densities on cobbles in varial zones averaged 42-percent of that of the permanent inundated areas. Further study by Blinn et al. (1998) found that periodically exposed zones in the Colorado River had 10-fold lower productivity than permanently wetted areas.

Gislason (1985) concluded that the effects of power peaking adversely influenced insect density along the margins of the Skagit River, Washington. Under fluctuating flows, insect density increased in the direction from shallow to deep water, and density decreased with increasing number of hours of dewatering prior to sampling. Diversity appeared to increase with water depth, and decrease with increased duration of dewatering. Like the previously mentioned studies, Gislason (1985) determined that the density of chironomid larvae was greatest in shallow water, suggesting that they are tolerant of stream margin dewatering.

In the Kootenai River, Perry and Perry (1986) reported that longer periods of high discharge between periods of fluctuation or low flows permitted the recolonization of the varial zone. Rock baskets deployed at high, intermediate, and minimum flow levels of the varial zone for colonization revealed the highest numbers of invertebrates in baskets located at the minimum flow levels, and thus within those that continuously remained inundated. However, they also observed that reductions in discharge immediately prior to benthic sampling resulted in the concentration of large numbers of insects near the water's edge, suggesting the establishment of a macroinvertebrate community that can move with the retreating shoreline. Examination of the zones of exposure in the Flathead River did not appear to be recolonized to a considerable extent, due to more frequent flow fluctuations (Perry and Perry 1986).

Other studies on the lower Flathead River have demonstrated that BMI production in the varial zone is severely limited due to daily dewatering (Hauer et al. 1989; DeVries et al. 2001). Hauer et al. (1989) found that the relatively frequent fluctuations in discharge below Kerr Dam favored mid-water emerging species, including members of the Hydropsychidae, Baetidae, and Ephemerellidae families. This was noted particularly in sections with a wide varial zone. There appeared to be a selective pressure against shoreline-emerging insects such as plecopterans (Hauer et al. 1989). DeVries et al. (2001) found that BMI density in margin areas of the lower Flathead River was significantly lower relative to the community in mid-channel areas. Benthic taxa richness was also lower, but the difference was not always statistically significant. The benthic fauna in margin areas contained a much higher percentage of snails, aquatic earthworms, and chironomids than mid-channel habitats.

5.1.4.1.2. Drift Responses

Many studies have observed and reported on the drift response of macroinvertebrates to flow regulation, both in regulated rivers and artificial streams. White and Wade (1980) evaluated a proposed increase in daily fluctuating flows on a 28-mile reach of the South Fork Boise River, Idaho, below Anderson Ranch Dam during the winters of 1977 and 1978. They examined the effects of progressively increasing the existing maximum discharge from 1600 cfs, increased from the base 200 cfs at a ramped rate of 35 cfs every 10 minutes, to 2400 cfs, with an upramping rate of 140 cfs every 10 minutes. Catastrophic drift occurred at all discharges tested, with drift densities increasing with the faster rates of change and the higher maximum discharges. White and Wade (1980) reported that drift was dominated by Chironomidae, *Baetis*, and *Ephemerella*, and that drift occurred as the increased discharge reached each site, but the behavioral drift was also increased during the night following the pulse increase.

Irvine and Henriques (1984) monitored a flow experiment similar to that reported by White and Wade (1980). A base discharge of 530 cfs was maintained for one month, before rates of discharge were manipulated over a 5-day period. On the first and fifth days, the base flow was maintained. On the second day, flows were increased at a rate of 106 cfs per hour over four hours to a peak of 1,059 cfs, and then decreased at the same rate. The following third and fourth days used rates of 317 cfs and 3,708 cfs, for peaks of 2,118 cfs and 3,708 cfs, respectively. Increased flows caused catastrophic drift at all three levels, with drift densities peaking shortly before maximum discharge was attained. Drift densities also tended to lessen by the third day of experimentation. Drift was dominated by oligochaetes, chironomids, and the caddisfly *Oxyethira albiceps*.

Many other studies involving operating hydroelectric dams report similar observations of increased drift following peak flow releases. Céréghino and Lavandier (1998a, 1998b) found that both Plecoptera and Ephemeroptera taxa displayed catastrophic drift at the beginning of daily peaking increases that ranged 3-10 times the base flow, in addition to the diel behavioral drift seen naturally upstream of the peak releases. The nemourid stonefly *Protonemum beatensis* and the mayfly *Baetis alpinus* were the dominant drifting taxa. Mayfly drift was greatest in the fall, when the differences between natural and peaking flows were greatest. Perry and Perry (1986) reported increased drift densities for many taxa during increases in discharge below Hungry Horse Dam on the Flathead River, Montana, but found drift densities decreased after the first hour of the test. They also compared the effects of two different rates of increase in discharge on drift. They found that a direct increase in discharge from 144.8 to 9,993 cfs did not cause more drift than a stepped rate from 144.8 to 3,400 to 9,993 cfs, although the faster rate did tend to entrain more terrestrial insects from the shoreline areas.

Moog (1993) found that pulse releases below several Austrian dams of 30- to 60-times the minimum flow increased the BMI drift at the beginning of pulses and during their recession. He also estimated that artificial flow fluctuations reduced the BMI biomass 75-95-percent within the first few kilometers from the dam, when compared to undisturbed areas. A reduction of 40-60-

percent of biomass was detected within the following 20-40 km, indicating the amount of influence flow fluctuations can have upon the downstream communities (Moog 1993).

Several studies using experimental streams have examined the drift response to both decreases and increases in flow (Poff and Ward 1991; Imbert and Perry 2000; Corrarino and Brusven 1983; Irvine 1985). Corrarino and Brusven (1983) used experimental concrete channels and subjected them to a reduced flow schedule in a series of tests. An initial reduction in flow by one-half caused a minor pulse in drift, but a second, more substantial reduction clearly caused an increase in catastrophic drift, with pulses of drifting insects apparent as soon as one hour after flow reduction was complete. Two taxa were of interest during the second reduction because of their differing responses. The most abundant organism in drift, *Baetis tricaudatus*, displayed a delayed catastrophic response, waiting until midnight, whereas *Simulium* sp., which showed no consistent drift trend in the control channel, responded immediately, entering the drift within an hour of the second reduction. The greatest drift pulse for all experiments occurred in darkness, generally at midnight.

In a contrasting study, Irvine (1985) examined the effect of sudden and successive increases in flow on BMIs. After maintaining a constant discharge in 2 experimental streams (treatment and control) for several months, flow in his treatment stream was increased five-fold for 3 successive pulses, each lasting 45 minutes with 45-minute low-periods in between pulses. These series of pulses were repeated on four successive days each week for three weeks. Benthic macroinvertebrate drift density (comprised mostly of chironomids, oligochaetes, and the caddisfly *Oxyethira albiceps*) increased significantly during the first pulse of the study, with the increase lessening with each successive pulse. By the end of the 3-week study, changes in discharge did not cause changes in drift. Irvine (1985) postulated that drift densities decreased because fewer invertebrates were available to drift, as benthic densities also declined significantly by the conclusion of the study. Decreased benthic densities were partially attributed to loss to drift, but some taxa possibly retreated deeper into the hyporheos to avoid the flows, thereby making them less easily captured by the sampling device used.

Imbert and Perry (2000) conducted two tests in experimental streams to assess BMI responses to stepwise and abrupt changes in non-scouring flow. In the stepwise experiment, flow was incrementally increased every 30 minutes to reach a two-fold increase in discharge that was held for 2 hours, before being returned to the base-flow condition in similar, stepwise decreases. After 22 days at base flow conditions, flow was abruptly increased 2.4x for 3 hours, and then abruptly returned to base-flow levels. Relative to reference streams, both experimental increases resulted in an increase in drift richness and drift density. Peaks in drift richness and density were reached immediately in the abrupt experiment, but occurred between the second and fifth flow increase in the stepwise experiment. Relative to reference streams, which maintained base-flow conditions, the percentage of total benthic invertebrates in the drift increased 10-times in the stepwise experiment and 33-times in the abrupt experiment, suggesting that abrupt increases in flow had a stronger effect on BMI drift than did stepwise increases.

Bond and Downes (2003) used a series of eight artificial streams to investigate the effects of increases in flow and associated suspended sediments on BMI fauna. While the addition of sediment alone had little effect on the BMI community, flow increases caused large increases in drift density and richness. In the first experiment, flows were increased from 2.5 L/s (0.0883 ft^3/s) to 12.5 L/s (0.441 ft^3/s) in four 20-minute intervals, whereas in the second experiment, the number of flow increases was halved, resulting in a steeper upramping rate. Results showed drift densities for the second experiment were 6-times that of the first experiment and drift richness in the second experiment was 2.4-times that of the first experiment. However, the investigators did not compare results between the two experiments, so it is unclear whether these differences in drift responses were due to the differing rates of flow increase or due to seasonal differences.

A series of studies on the River Oriège (Pyrénées,France) investigated the impact of hydropeaking on macroinvertebrates, both overall and for specific taxa (mayflies, stoneflies) and found that downstream from the powerplant, most of the larvae were flushed at the beginning of the peak flow, foregoing the diel patterns normally seen in the river (Céréghino and Lavandier 1998a, 1998b; Céréghino et al. 2002; Céréghino et al. 2004). These studies all noted that the disturbance was even higher during seasonal periods when amplitudes between natural flow and peak flow were greater, and that the lowest densities and biomasses were immediately downstream from the power plant, gradually increasing in abundance with increasing distance downstream from the dam, thus emphasizing the role catastrophic drift can play influencing benthic macroinvertebrate communities.

5.1.4.1.3. Stranding

When flows are reduced rapidly, as in flow regulation or with the recession of natural floodwaters, many previously inundated organisms are able to return to the main channel, but a substantial number can become stranded (Perry and Perry 1986; Rempel et al. 1999; Robinson et al. 2004). Stranding of macroinvertebrates occurs mostly due to rapid flow reductions along gently sloping shorelines. Depending on the season, time of day, and the length of period that the shoreline remains exposed, stranded organisms can dry out and die. Downramping, or the process of rapid flow reduction from a regulated system, can dramatically affect the degree of impact imparted to the aquatic biota based upon the rate at which these areas become exposed. Many hydroelectric projects today must follow prescribed downramping rates when making flow adjustments. Importantly, a number of studies have shown that the faster the downramp rate the greater the risk of stranding and trapping of invertebrates or other aquatic taxa (Brusven and MacPhee 1976; Brusven et al. 1974). Cushman (1985) summarized major effects of rapidly varying flows on aquatic biota generally, while Hunter (1992) did so with a focus on salmonids. Both acknowledged the importance of rate of stage change and the associated effects of stranding and trapping.

During the series of experimental flow reductions in the Snake River, Brusven et al. (1974) determined the fate of stranded organisms. The initial flow reduction from 27,000 to 18,000 cfs resulted in the stranding of chironomids, the principal inhabitants of the shoreline at that level. Greatest stranding occurred during the reduction from 12,000 to 7,700 cfs. The mayflies *Baetis* and *Ephemerella*, and chironomids were most readily observed as stranded, and substantial numbers of *Hydropsyche* sp. were found seeking shelter under rocks and in algal mats.

Some macroinvertebrates survived long periods of dewatering, mostly by successfully finding moisture in *Cladophora* mats or under rocks. Brusven et al. (1974) noted that chironomids demonstrated remarkable survival after 96 and 120 hours, and Trichoptera and Lepidoptera showed high survival after 48 hours of exposure. Mayflies were found to be most sensitive, with high mortality after 24-48 hours of exposure. High survival was partially attributed to the cooler air temperatures at the time of flow reductions (March). The 96-hour exposure period was marked by higher air temperatures (85º F), which increased mortalities to or beyond 50-percent for most mayflies, caddisflies, and lepidopterans. In other cases, stranding and desiccation results in extensive mortality as in the study by Matthaei and Townsend (2000) in the Tye Burn, New Zealand where invertebrate density was reduced 50 percent or more by losses due to stranding. The intensive studies of benthic community responses on the Spöl River also identified the stranding of invertebrates in shallow riparian areas as floodwaters receded as an important source of post-flood mortality (Robinson et al. 2004).

The effect of seasonality was also seen in the flow reduction tests conducted by Corrarino and Brusven (1983). In the spring, the initial reduction of flow by one-half exposed only 8-percent of the wetted perimeter, stranding few macroinvertebrates. The second, more substantial reduction exposed 31-percent of the wetted zone, stranding many macroinvertebrates; however, more than half of those stranded managed to reach the water again. In the fall, both levels of flow reduction resulted in nearly 100-percent stranding. Higher spring survival was attributed to cooler temperatures and the presence of larger, more mobile mature instars during that time. Most insects stranding in the spring were *Rhithrogena hageni*, Chironomidae, and *Simulium* sp whereas taxa stranded in the fall were largely Chironomidae and Hydropsychidae. The mayfly *Baetis tricaudatus*, most prevalent in drift during the experiments, was successfully able to avoid stranding in all tests.

Perry and Perry (1986) observed that a return to minimum flows after higher sustained flows in the Kootenai River below Libby Dam in Montana resulted in significant losses of BMIs. They also found that more stranding occurred during a faster rate of decrease in flows. In January and March 1981, abrupt flow decreases from 19,986 cfs to 8,016 cfs in 10 minutes resulted in greater stranding (50,000 to 100,000 individuals/m^2), than in February when flows were downramped to 8,016 cfs at a normal winter rate of 2.2 vertical feet per hour. Stranded organisms were mostly comprised of Chironomidae, Simuliidae, *Ephemerella inermis*, and *Baetis tricaudatus*. These findings agree with Corrarino and Brusven (1983) regarding the dipterans, but are contrary for *Baetis tricaudatus*.

5.2. Temperature

Ward and Stanford (1979) suggested that the type of thermal modification following hydroregulation through dams largely depends on the release depth, thermal stratification of the impoundment, retention time, and dam operation. They considered six categories of thermal modification, which include increased diurnal constancy, increased seasonal constancy, summer depression, summer elevation, winter elevation, and thermal pattern changes. Sinokrot et al. (1995) found that dams affected the thermal regime differently depending on whether water was drawn from the surface (epilimnetic) or deeper (hypolimnetic) portions of the reservoir. Epilimnetic releases often cause increased summer temperatures, which can be detrimental to the biota in native cold-water streams (Ward and Short 1978; Zimmerman and Ward 1984). In contrast, hypolimnetic releases moderate diurnal and seasonal fluctuations, leading to warmer winter temperatures, and cooler summer conditions (Ward and Stanford 1979; Armitage 1984; Petts 1984; Sinokrot et al. 1995).

The effects of thermal regulation on aquatic organisms from dam operations vary widely (Murchie et al. 2008), but are well documented in the literature. A regulated thermal regime may cause shifts in the trophic food web (Wootton et al. 1996), can lead to changes in community structure (Vinson 2001), and in severe cases can lead to extinction of species (Zhong and Power 1996). Organisms in rivers are well-adapted to the thermal dynamics in their native environment, andchanges in these dynamics can have deleterious effects on organisms (Steel and Lange 2007). For example, benthic communities below hypolimnetic releases generally have a less diverse fauna, with losses of many mayfly, stonefly, and caddisfly taxa (Brittain and Saltveit 1989; Hauer et al. 1989; Stanford and Ward 1989; Munn and Brusven 1991). Stabilized, warmer winter temperatures can eliminate the thermal cues used by many taxa to break their egg diapause, whereas cooler summer temperatures may disrupt the life histories of the taxa present by reducing the number of available degree days needed to complete development, reducing the growth rate or fecundity of certain species, or delaying or extending the emergence period (Ward and Stanford 1979; Armitage 1984).

Regulated thermal regimes can alter components of diversity because some species flourish in the regulated environment whereas others decline. For example, Ward and Voeltz (1988) reported high densities of chironomids, simuliids, and *Hydra*, but very few mayflies, stoneflies or caddisflies below a Colorado dam. In contrast, species not found below the dam were found in expected numbers above the dam; thus, species diversity had been greatly reduced in the regulated thermal regime. Vinson (2001) found that an extensive amphipod population below Flaming Gorge Dam in Utah was due, in part, to low reproductive success of aquatic insects resulting from the altered thermal regime. Again, the location from which flows are released may lessen the effects of regulation on thermal regime and species assemblage, as several studies have shown increased densities of *Baetis*, *Ephemerella*, and *Tricorythodes* mayflies below hypolimnetic release dams (see Brittain and Saltveit 1989).

5.3. Substrates

As discussed in Section 3.3, substrate and flow interact greatly to determine what habitat is available to macroinvertebrate communities in a stream or river. Where water is slow moving or still, substrate may be relatively homogenous; where currents are active or flow is swift and turbulent, more diverse and heterogenous substrates will occur (Minshall 1984; Allan and Castillo 2007). Therefore, the effect of regulation on flow will dictate the substrate conditions below hydropower projects, such as was explained for reduced and increased flows (Sections 5.1.2, and 5.1.3).

In terms of this flow-substrate interaction, this section will summarize the effects of different flow release levels, or regimes, on substrate mobility, and the possible effects on the benthic community. The discussion focuses on the mechanics of gravel bed streams generally, and specifically with respect to three different flow releases (Figure 5.3-1) that do not exceed the bankfull flow condition (Schmidt and Potyondy 2004). Flows would not usually exceed bankfull flow in regulated streams, except when inflows are too high to be regulated and excess flows are passed downstream as spill.

Flow releases under a **low flow** regime would not be sufficient to entrain and transport fine sediments from the surface of the channel bed (Figure 5.3-1). Under unregulated conditions, the low flow regime would occur during much of the winter period on glacially-fed rivers. During these low flows, there would be little/no transport of sediment, ergo little change to the structural habitat of BMI.

Flow releases under a **medium flow** regime would be sufficient to transport fine sediments on the surface of the channel bed, but not sufficient to initiate mobilization of the armor layer (Figure 5.3-1). As flows increase, eventually a condition is reached where the river is capable of entraining and transporting fine both sediments and BMI from the surface of the armor layer of the channel bed. Under unregulated conditions, the medium flow regime would likely occur during spring snowmelt, and occasionally during summer and fall periods as well. These flows are capable of transporting fine sediment from the surface of the armor layer. Milhous (1998) refers to this process as surface flushing of fines. The primary effects of PTF in the medium flow range would consist of entrainment of BMI from the channel bed surface and downstream transport of the organisms as drift. The transport of macroinvertebrates would be associated with surficial flushing of fine sediments from the surface of the armor layer. High frequency PTF in this range could theoretically result in a continuous suspension (drift) – deposition cycle that may affect BMI community density and diversity.

Flow releases under a **high flow** regime would be sufficient to mobilize the armor layer of the substrate and potentially alter the shape of the channel (Figure 5.3-1). Pulse flows with a magnitude in this range would be very disruptive to the BMI community. The gravel matrix structure at the surface of the channel bed, which normally would provide shelter for the macroinvertebrates, would be disrupted. Some of the macroinvertebrates would likely seek refuge deeper within the gravel matrix (hyporheic zone), while others would be crushed or swept downstream. High frequency PTF of this flow magnitude could completely eliminate BMI communities in sections of stream where substrates are continually disrupted and physically disturbed. Similarly, when flows are decreased from a high flow condition, eventually the coarser sediment settles on the channel bed and the channel morphology stabilizes. As flows are decreased even further, the finer sediments and BMI within the drift would settle on the surface of the channel bed.

In addition, the establishment of a dam and impoundment of the reservoir behind it will inevitably affect sediment transport and supply in the river downstream. Depending on the size of the reservoir, large amounts of sediment will be trapped, with only a portion of the former load released below the dam into the downstream reaches (Hay 1994; Ibáñez et al. 1996; Brandt 2000; Anselmetti et al. 2007). Large dams trap virtually all of the sediment delivered into the reservoir from upstream sources, whereas smaller dams can have more variable trap efficiencies, ranging from 10 to 90-percent or higher (Brune 1953; Williams and Wolman 1984; Grant et al. 2003). In addition to the reduced sediment load and concentration, the grain size of released sediments will also be decreased to downstream reaches (Brandt 2000). The lack of sediment transport downstream of the dam introduces the problem of "hungry water," clear water released from the dam with excess energy to move sediment, but with no sediment to move (Kondolf 1997). This energy is instead expended on erosion of the channel bed and banks downstream, resulting in channel incision and bed armoring until equilibrium is reached and the material cannot be moved by the flows (Williams and Wolman 1984; Kondolf 1997). Reservoirs that reduce flood peaks downstream may also potentially reduce the effects of hungry water, which can result in channel shrinking, or allowing increased sedimentation. Any of these effects will have an influence on the substrate composition downstream of a dam, and in turn on the benthic communities.

5.4. Turbidity

Because varying amounts and sizes of sediment are trapped behind a dam, turbidity levels are often lowered in reaches downstream (Ward 1976; Baxter 1977; Ward and Stanford 1979; Lowe 1979; Armitage 1984; Stevens et al. 1997). The decreased turbidity increases light penetration, and favors increased submerged algal and macrophyte production (Baxter 1977; Lowe 1979). Dense growth of algae and macrophytes have been reported in reaches below dams that supply constant flow, stable substrates, clear water, and ice-free conditions. This amelioration of harsh environmental conditions can encourage greater biotic interactions often enabling competitive dominance by a few species with an overall decrease in diversity (Zimmerman and Ward 1984; Rader and Ward 1988; Ward and Voelz 1988; Stevens et al. 1997; Vinson 2001).

Despite the possibility that reservoirs trap most of the sediment, suspended sediments are not trapped, thus, waters downstream of dams are not necessarily clear. Sediment is often resupplied to the channel from tributaries, hillslopes, and channel erosion (Ward and Stanford 1983a; Ward and Stanford 1995; Brandt 2000; Vinson 2001). Anselmetti et al. (2007) reported on the effects of particle transport and lake sedimentation for several reservoirs in a hydropower system on River Aare in Switzerland. The upper three reservoirs in the system receive on average a total of 271 kilotons (kt) of sediment per year, trapping all but 39 kt/yr of the fine fraction (4 mm in diameter or less) which is transported downstream to the River Aare, which ultimately leads to another reservoir, Lake Brienz. In glacial systems, ultra-fine clay particles may not have sufficient residence time in reservoirs to fully settle out, and can be released through the dam to downstream reaches. Finger et al. (2006) estimated that only about 3 percent of the sediment trapped in the reservoirs in the River Aare system is fine matter (less than 4 mm diameter), and therefore the load of fine matter transported downstream does not change a lot due to damming. The hydropower operations in that system, therefore, shift particle inputs from summer to winter. As a consequence, the River Aare becomes more turbid during winter and less turbid during summer, influencing the seasonal light regime and subsequently the dynamics of algal growth, as proposed by Uehlinger et al. (2002) (see Section 2.2).

5.5. Riparian Zone and Organic Matter Contributions

The construction of dams and the resulting regulation in flows has resulted in dramatic shifts in riparian ecosystems across a broad spectrum of river systems worldwide (e.g. Tabacchi et al. 1998; Merritt and Cooper 2000; Johnson 2002; Watkins et al. 2010; Greet et al. 2011; Kennedy and Ralston 2012). As mentioned above, regulation of river flows can reduce peak flood discharge or alter its timing (Petts and Gurnell 2005), reduce overall downstream discharge, and alter geomorphic processes such as sediment transport and deposition (Jansson et al. 2000; Nilsson and Berggren 2000), typically starving downstream reaches of sediment impounded behind dams. In the lower reaches of regulated rivers, flow control typically results in floodplains being inundated less frequently and for shorter duration.

River regulation can result in a decrease in the gross quantity of allochthonous inputs to aquatic habitats (Richardson 1992; Benfield 1997). Floodplains are thought to serve as a reserve of organic matter which is periodically incorporated into the riverine food web during periods of ecosystem expansion, which is the fundamental insight of the Flood-Pulse Concept (Junk et al. 1989) (see Section 2.1). Rates of incorporation of terrestrial matter may be reduced in large river floodplains under flow regulation as the entrainment of stores of partially decayed organic matter is restricted by reductions in overbank flows (van der Nat et al. 2002; Doering et al. 2007). Hydroperiod and flow dynamics can also have a dominant effect on the overall ecosystem rate of decomposition of terrestrial organic matter. Studies have shown that rates of terrestrial decomposition are notably slower than inundated environments (Ribas et al. 2006; Leberfinger et al. 2010; Dieter et al. 2011). Leaf litter accumulating across unregulated floodplains or riparian gravel bars is subject to repeated cycles of stranding and rewetting that may enhance its quality as a food resource for both macroinvertebrates (van der Nat et al. 2002; Riedl et al. 2013) and microbial communities (Dieter et al. 2013).

Although there is a strong positive correlation between the abundance of shredding invertebrates and subsequent processing rates for terrestrial organic matter from river systems around the world (Wallace et al. 1997; Whiles and Wallace 1997; Hutchens and Wallace 2002), the effects of river regulation on this relationship are less clear. Using a comparison above and below dams on 5 small streams in Spain, Mendoza-Lera et al. (2012) found a consistent 20-percent reduction in litter breakdown rates below dams. They attributed this decline to reductions in invertebrate populations in stream reaches below the impoundments noting the specific decline of dominant shredders in the stonefly family Nemouridae, particularly the genus *Protonemura*. In contrast, Short and Ward (1980) documented increases in litter processing at a regulated Colorado River site compared to an unregulated tributary and hypothesized that the "winter warm" thermal regime that existed below the dam enhanced microbial processing which served to compensate for the virtual absence of shredder species.

One of the most predictable effects of the regulation of rivers is the encroachment of terrestrial vegetation into channels (Jansson et al. 2000; Bejarano and Sordo-Ward 2011; Miller et al. 2013), a process which is hastened by greater reductions in flow (Poff and Zimmerman 2010). The total area of terrestrial vegetation has been found to increase downstream of impoundments (Johnson 1997; Nilsson et al. 1997; Merritt and Cooper 2000; Mortenson and Weisberg 2010), as low flows expose previously inundated substrates, promoting shifts from broadleaf to coniferous dominance of floodplains. Even with projected expansion in the total area of riparian forest, quantities of organic matter inputs from coniferous vegetation are known to be reduced compared to those typically observed from deciduous dominated riparian zones and floodplains (Maloney and Lamberti 1995; Lennon and Pfaff 2005; Richardson et al. 2005) in addition to being more evenly distributed in time (in contrast to the characteristic fall peak for inputs from deciduous taxa).

This, in turn, may change the quality of organic matter inputs to the river itself (Snyder et al. 2002), generating a possible long-term decrease in the bioavailability of the leaf litter resources available to aquatic consumers (Richardson et al. 2004; Kominoski et al. 2011; Hoover et al. 2011). Aquatic invertebrate shredders can be influenced by leaf litter quality, selectively feeding on higher-quality leaf litter inputs (Graça 2001). Leaf litter that breaks down quickly in streams provides a readily available food source for shredders and is often correlated with high-quality leaf species (Irons et al. 1988, Cummins et al. 1989; Motomori et al. 2001). Feeding experiments have documented that nitrogen-rich species, such as alder, support higher shredder abundances than other lower-quality species, such as conifers (Irons et al. 1988; Motomori et al. 2001).

Riparian plant species have evolved in the context of particular flow regimes resulting in a range of morphological, life history and phonological adaptations (Merritt et al. 2010b). Reductions in flood peaks and frequency typically favor flood-intolerant species. Colonizing species of common riparian taxa such as *Alnus* spp., *Populus* spp., *Betula* spp., or *Salix* spp., may become less abundant as flow regulation constricts the active riparian zone. These taxa and similar colonizing species require high light and moisture and are generally intolerant of competition, and are further affected by decreases in new alluvium and associated river meandering (Johnson 2002; Johnson and Waller 2012). Overall decreases in plant species richness have been noted (Jansson et al. 2000; Merritt et al. 2010a) in reaches downstream from reservoirs.

Such change in riparian vegetative communities can facilitate the establishment of invasive species (Jansson et al. 2000; Nilsson and Berggren 2000; Merritt and Poff 2010; Mortenson and Weisberg 2010; Spellman and Wurtz 2011; Greet et al. 2012; Greene and Knox 2013). Riparian zones are particularly vulnerable to invasion because of their dynamic nature; although river regulation typically stabilizes hydrologic regimes, this represents a disturbance to the historic regime and may promote the establishment and spread of non-native species with greater competitive abilities in the absence of flood disturbances (Strom et al. 2012). When vegetative expansion is dominated by invasive species, additional complications and feedbacks between aquatic and terrestrial ecosystem function may occur. For example, the Russian olive (*Elaeagnus angustifolia*), an invader of riparian corridors through the American west, may alter stream organic matter dynamics by dramatically increasing litter inputs at the same time as it reduces light via shading, resulting in a net decrease in in-stream primary production (Mineau et al. 2012).

By displacing native species invasive plants can also reduce the species richness and overall quality of leaf litter inputs to the river available for stream consumers (Hood and Naiman 2000; Dangles et al. 2003; and Lecerf et al. 2007a). If an invasive plant differs from native vegetation in leaf litter chemistry, it can affect leaf litter processing by aquatic invertebrate shredders (Graça 2001). Aquatic microorganisms responsible for conditioning leaf litter often are more likely to colonize higher-quality species (Irons et al. 1988; Cummins et al. 1989; Lecerf et al. 2007a), making shredders, in turn, more likely to feed on properly conditioned leaf litter (Cummins et al. 1989). For example, Mineau et al. (2012) found that Russian olive litter decayed more slowly than native willow, thus substantial benthic storage of organic matter occurred as a consequence.

5.6. Nutrients

River regulation and the formation of reservoirs by impoundment affect aquatic ecosystems by disrupting transport, cycling, and ultimately, the amount and quality of energy and nutrients available to support primary and secondary production. A reduction in annual flow amplitude and the flood severity (Stanford et al. 1996) reduces the interaction of river and floodplains (see Flood Pulse Concept, Section 2.1), and can prevent lateral movement of organic matter and associated nutrients from floodplains to the river channel (Robertson et al. 1999). Where annual flooding is common and floodplain interactions are the natural condition, the overall effect in regulated rivers can be a greater contribution of autochthonous organic matter from algal production compared to floodplain soils or leaf litter sources (Robertson et al. 1999).

Reservoir formation can also change the nature of aquatic ecosystems by retaining river-derived POM, and exporting reservoir- derived POM, resulting in discontinuity in organic matter sources along river corridors (Ward and Stanford 1983a). Angradi (1994) demonstrated that phytoplankton and zooplankton from Lake Powell dominated the signature of POM below Glen Canyon Dam, and across major rivers in North America stable isotopes of C and N have been used to show that POM downstream of reservoirs is dominated by planktonic sources (Kendall et al. 2001). Coarse particulate organic matter (CPOM; > 1mm) is common in river systems and is often derived from terrestrial plants (Hedges et al. 1986; Thorp et al. 1998), whereas reservoir producers such as phytoplankton and zooplankton are typically smaller and comprise the fine particulate organic matter (FPOM; <1mm) size class. Perry and Perry (1991) found that FPOM dominated the POM in regulated river sections below dams on the Kootenai and Flathead Rivers, presumably due to the selective settling of CPOM in reservoirs. However, during hydropower generation, they observed an increase in CPOM from periphyton sloughing and resuspension of organic matter along river reaches. Primary production in reservoirs can also change the character of riverine DOM. Miller (2012) examined DOC along the Colorado River below Lakes Powell and Lake Mead and found that although DOC quantity decreased below reservoirs, its quality was higher and less aromatic. The authors suggested that either terrestrial-derived DOC was degraded in reservoirs or autochthonous DOC was produced.

The formation of reservoirs also affects the N and P along river corridors and disrupts the nutrient spiraling or the downstream movement of N and P between living and non-living components. A wealth of literature has examined how N and P are processed along river corridors (Peterson et al. 2001, Seitzinger et al. 2002, Mulholland et al. 2008). The overall retention of nutrients in lakes and reservoirs is positively correlated with N loading rate and retention time (Harrison et al. 2009). In general, N is more efficiently removed than P in reservoirs and lakes because inorganic N can be lost as N_2 gas via denitrification (see Figure 3.6-1) in the presence of anaerobic conditions and reduced carbon supply. Otherwise, N and P removal is mainly associated with particle settling and burial or retention in biota. However, in some cases reservoirs can act as net N sources in certain regions (Northeast US; Seitzinger et al. 2002) or with specific reservoir developments (Teodoru and Wehrli 2005).

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7. FIGURES

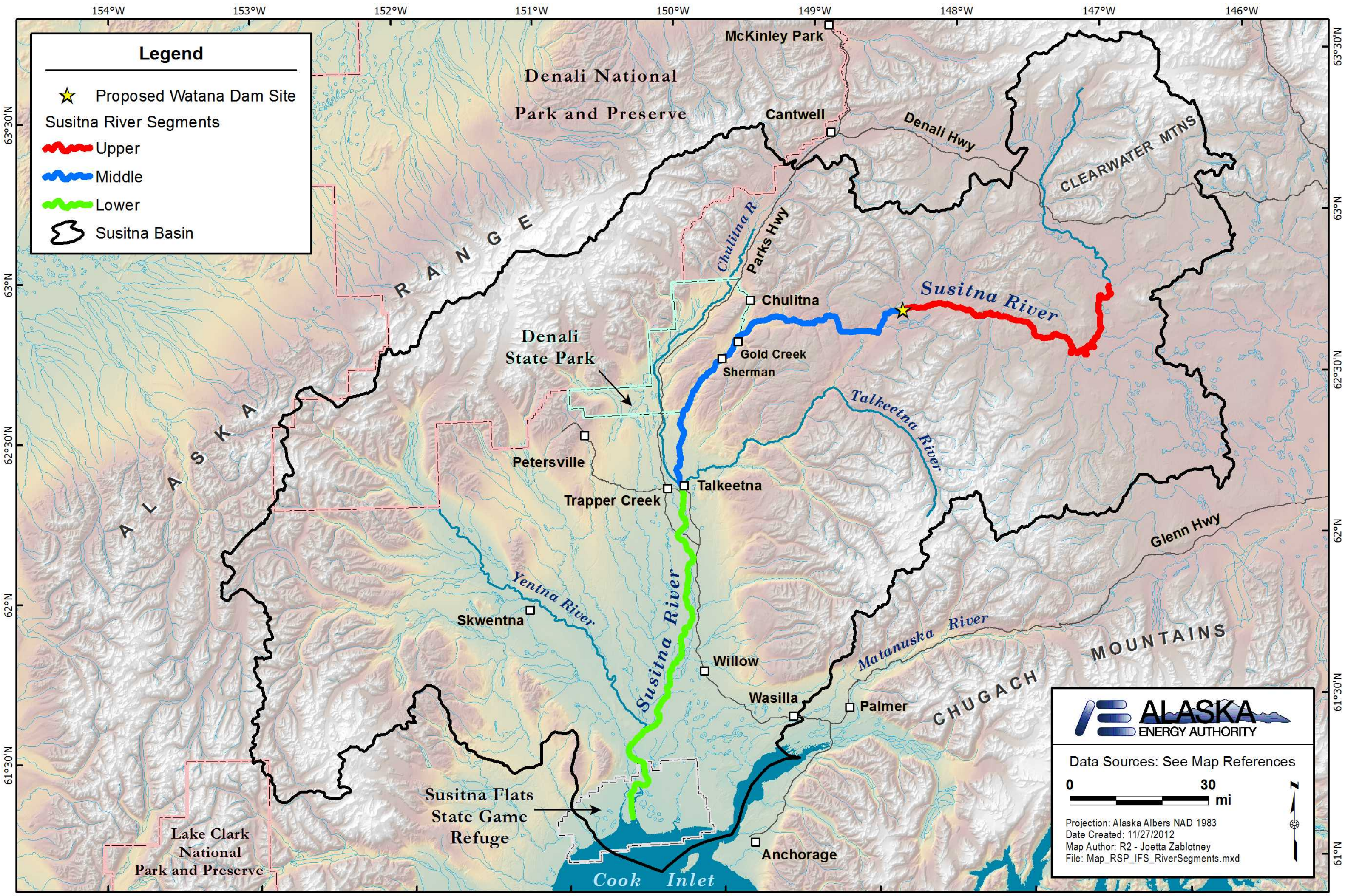
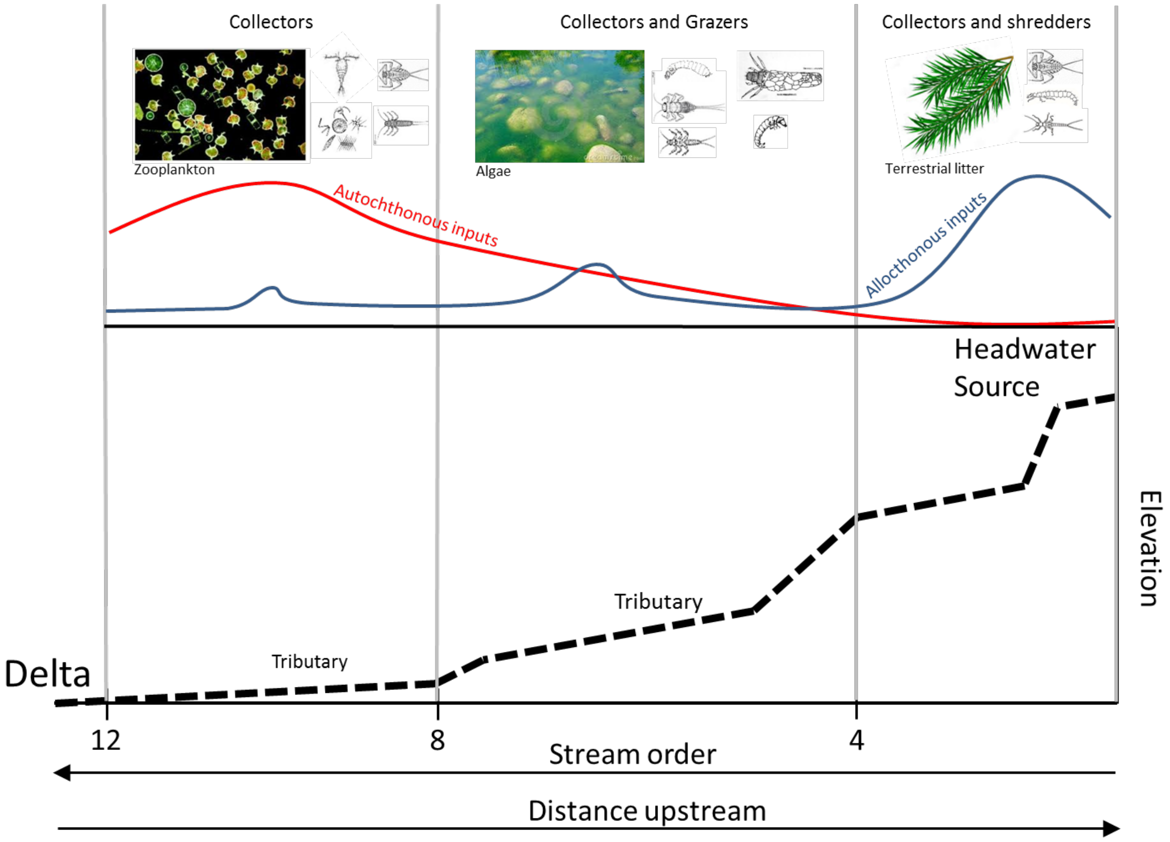


Figure 1-1. Susitna River Basin. Proposed dam site is located at the upstream end of the Middle Susitna River. Downstream from the dam site, the Susitna River receives unregulated inflow from tributaries. The largest downstream tributaries include the Chulitna, Talkeetna, and Yentna rivers.



Adapted from Vannote et al. 1980

Figure 2.1-1. Conceptual diagram of the River Continuum Concept, depicting the theoretical relationship between stream size, energy inputs, and ecosystem function (adapted Vannote et al. 1980).

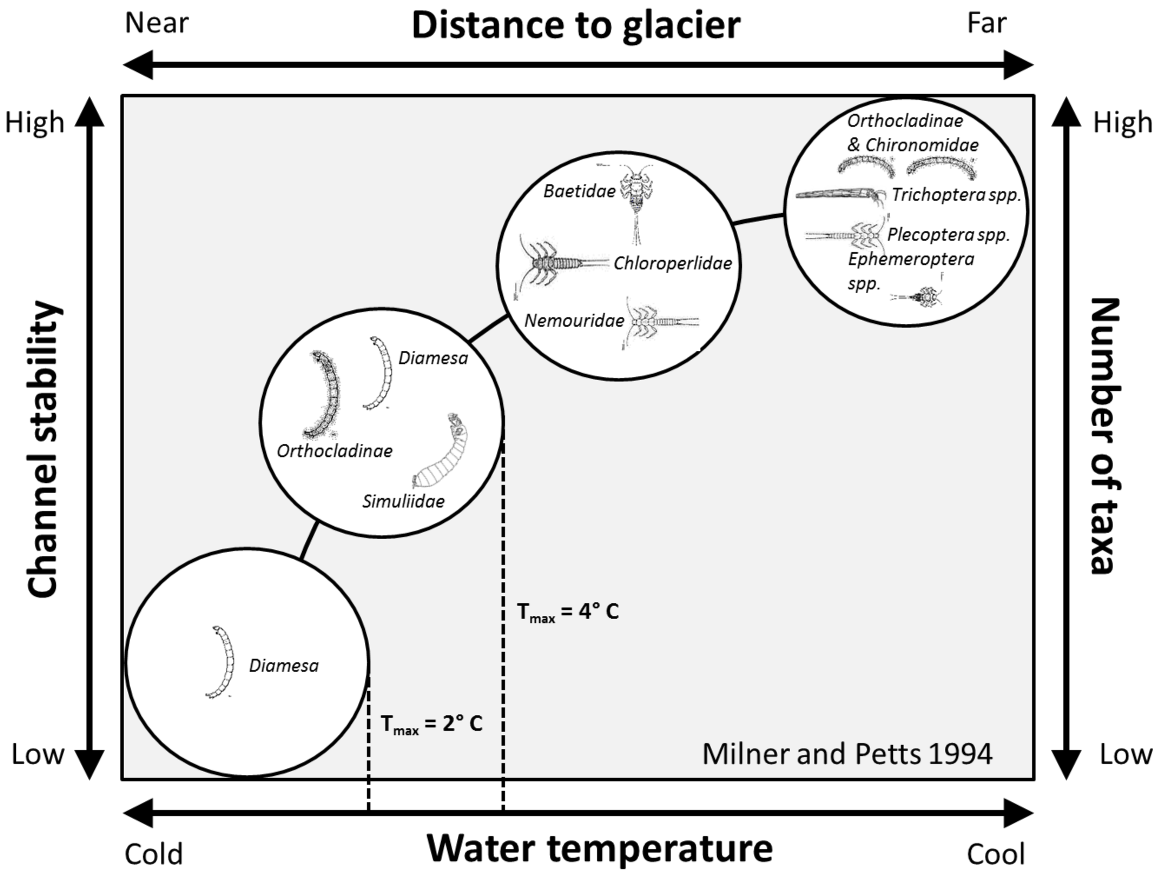


Figure 2.2-1. Points along a glacial river channel of increasing stability and water temperature where characteristic invertebrate taxa are typically added to the community. Redrawn from Milner & Petts (1994)

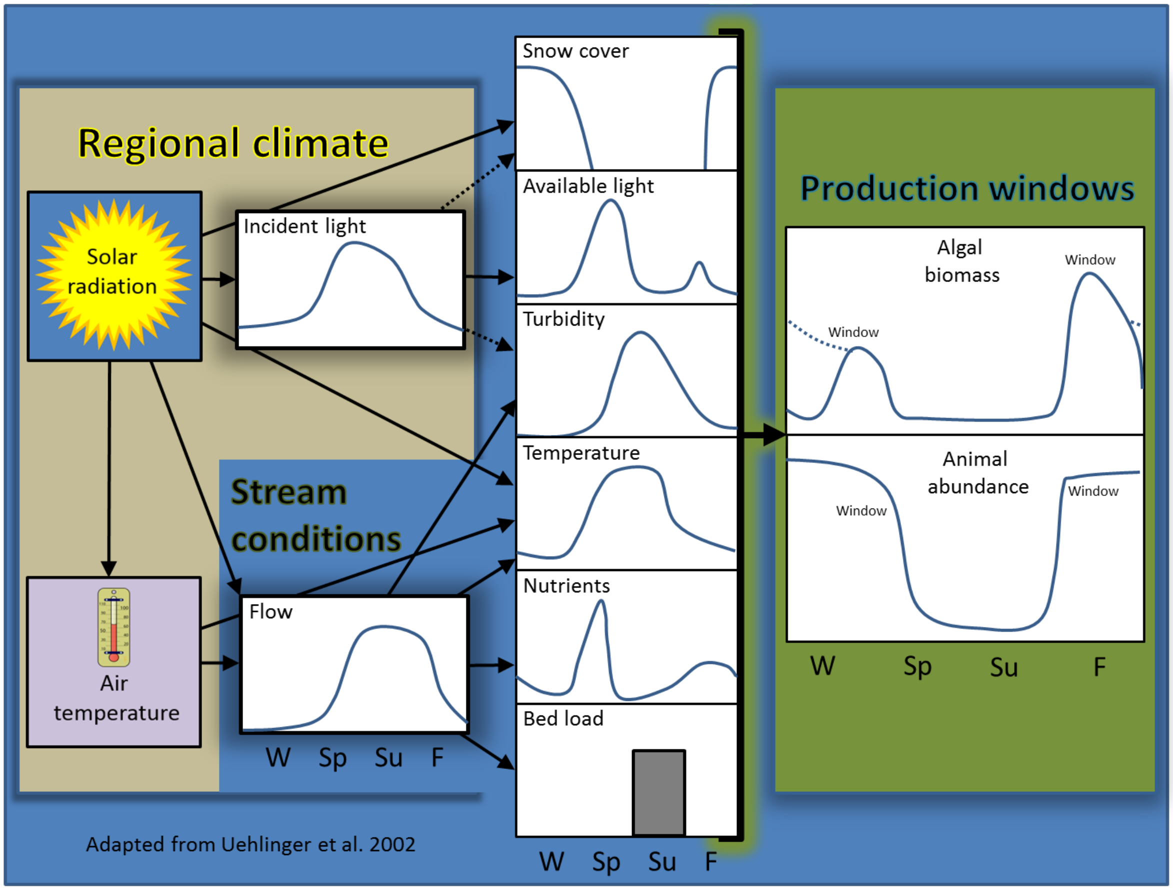


Figure 2.2-2. Conceptual diagram depicting the "windows of opportunity" for periphyton and benthic invertebrate production in the physico-chemical habitat template of glacial streams. Adapted from Uehlinger et al. 2002.

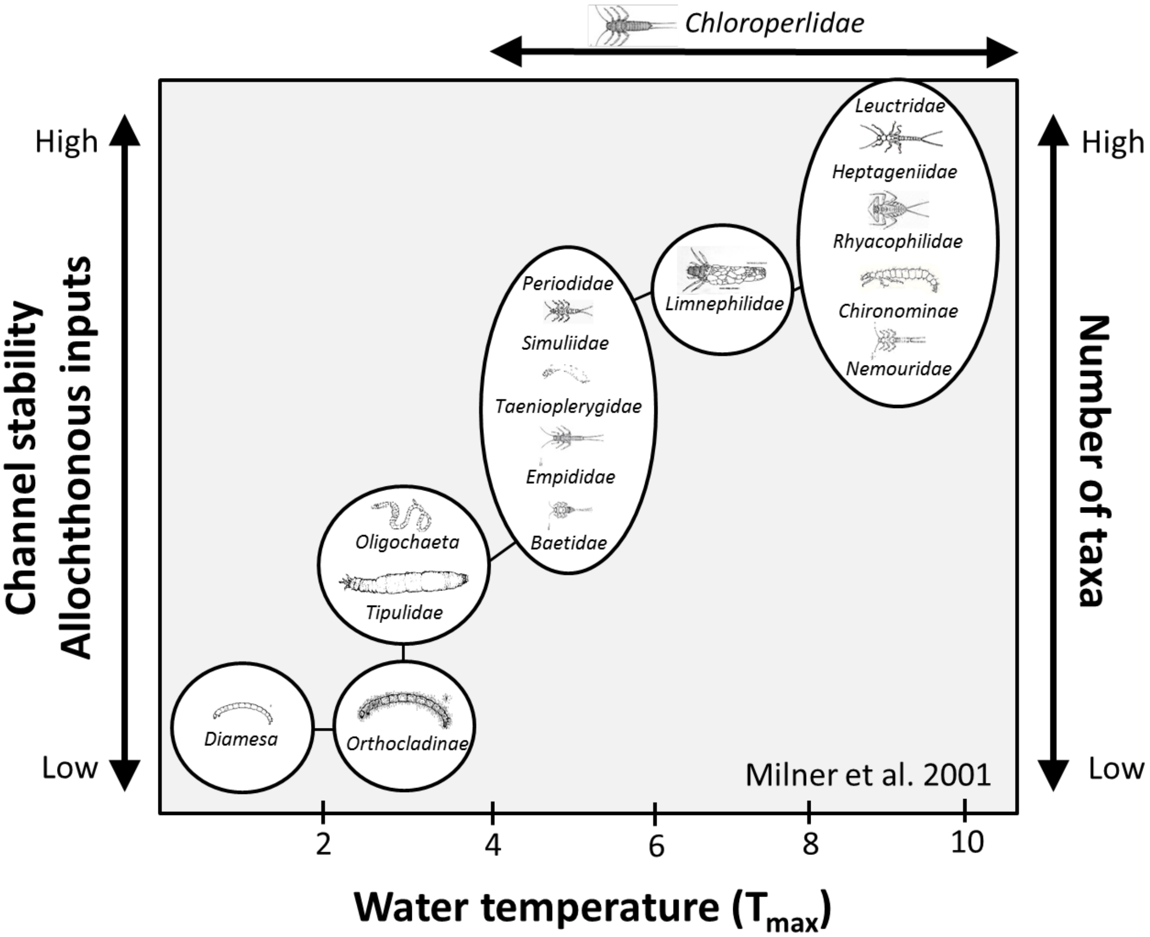


Figure 2.2-3. Modified conceptual model describing the likely first appearance of macroinvertebrate taxa along an upstream–downstream continuum from the glacier margin with increasing water temperature and channel stability during the melt season for European glacier-fed rivers. Adapted from Milner et al. 2001.

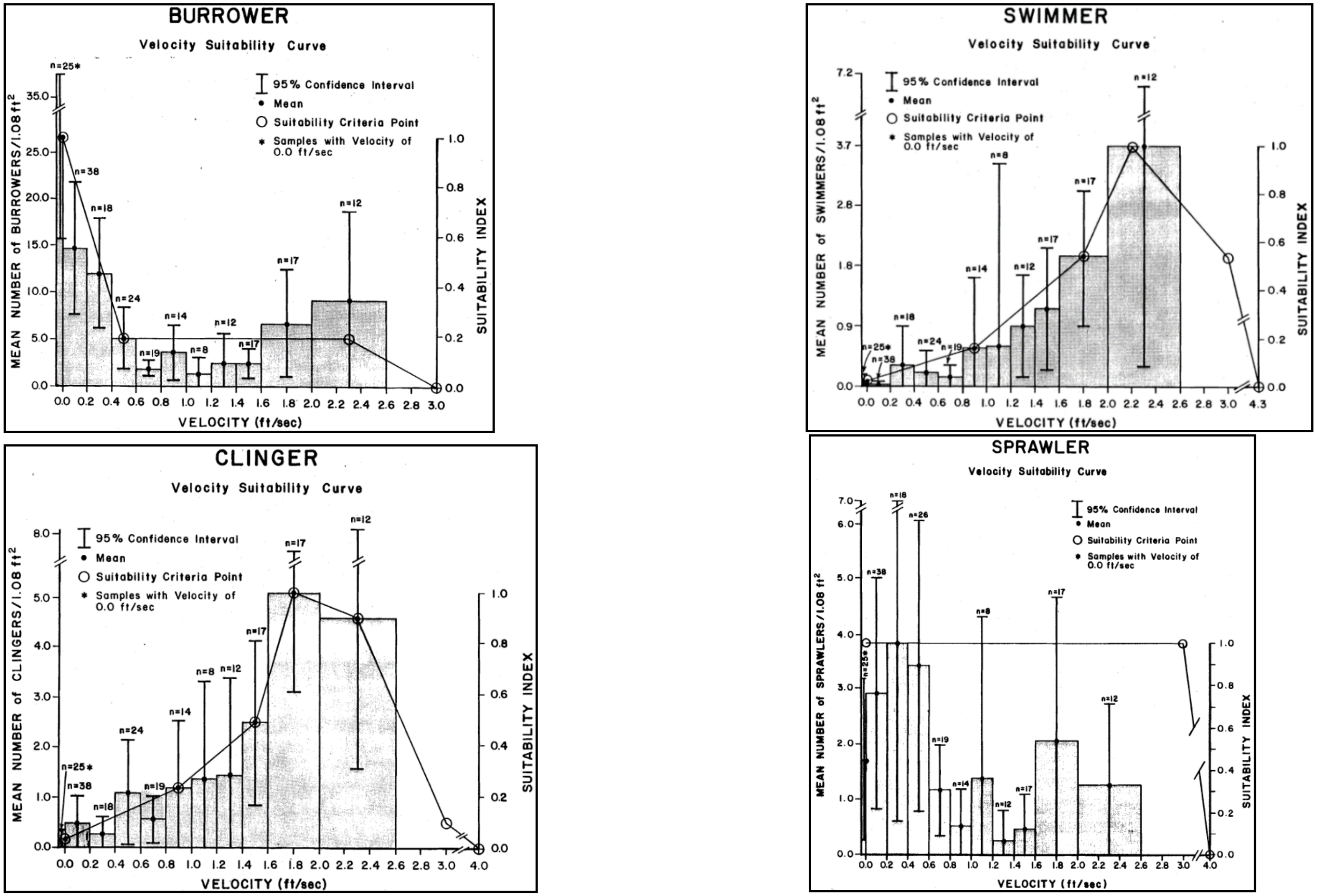


Figure 2.4-1. Velocity habitat suitability indices for benthic invertebrate guilds based upon sampling in the Middle Susitna River during 1984. Source: Hansen and Richards (1985).

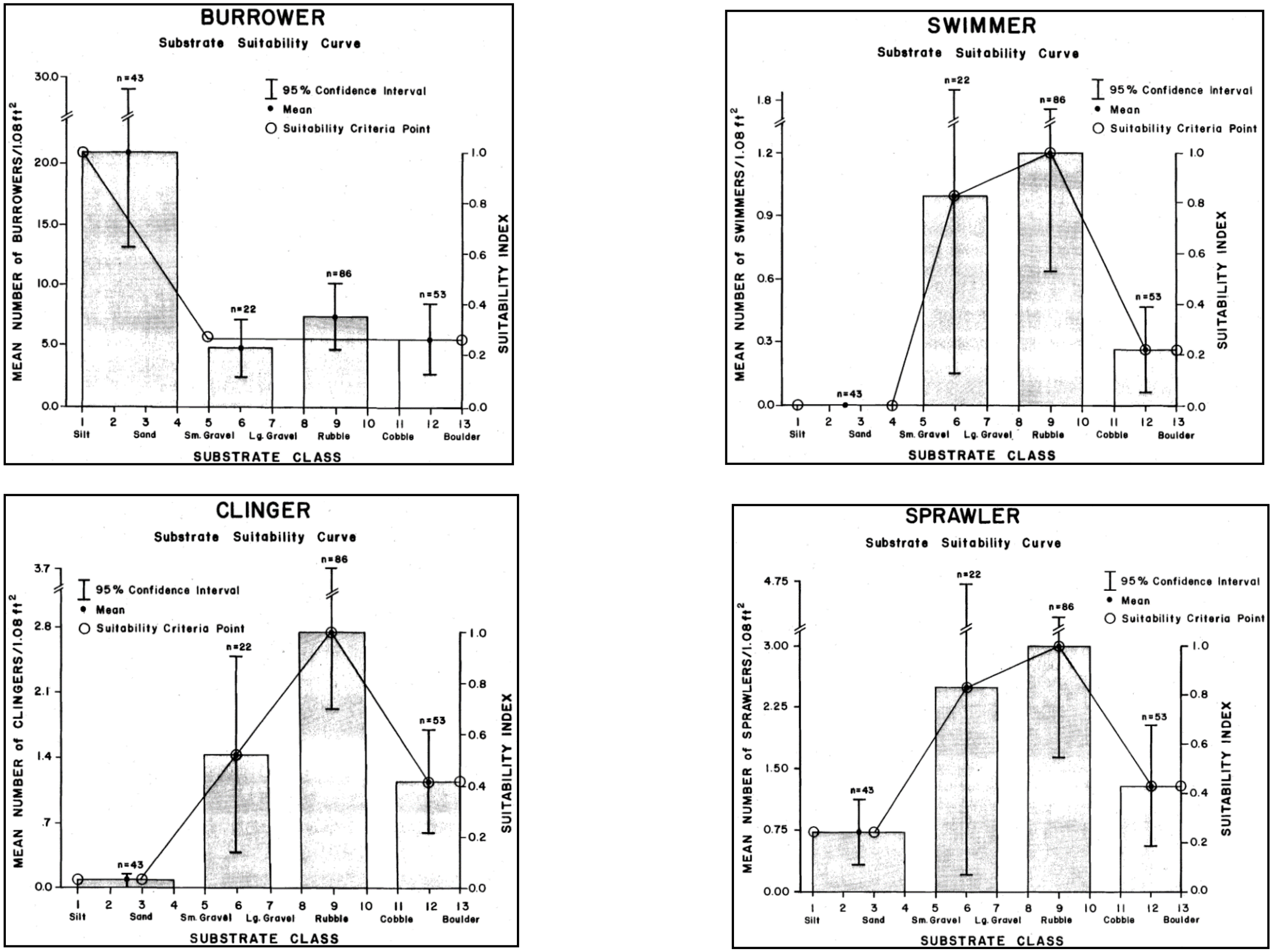
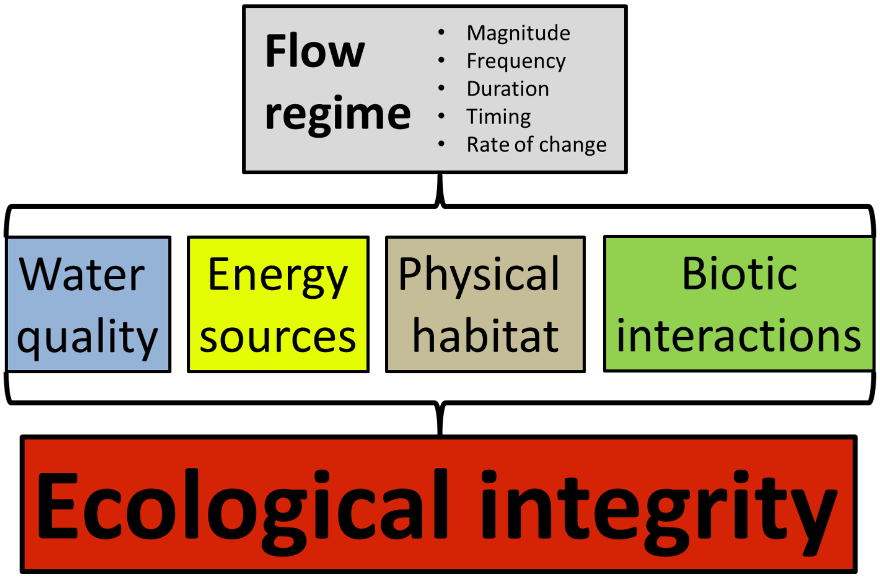
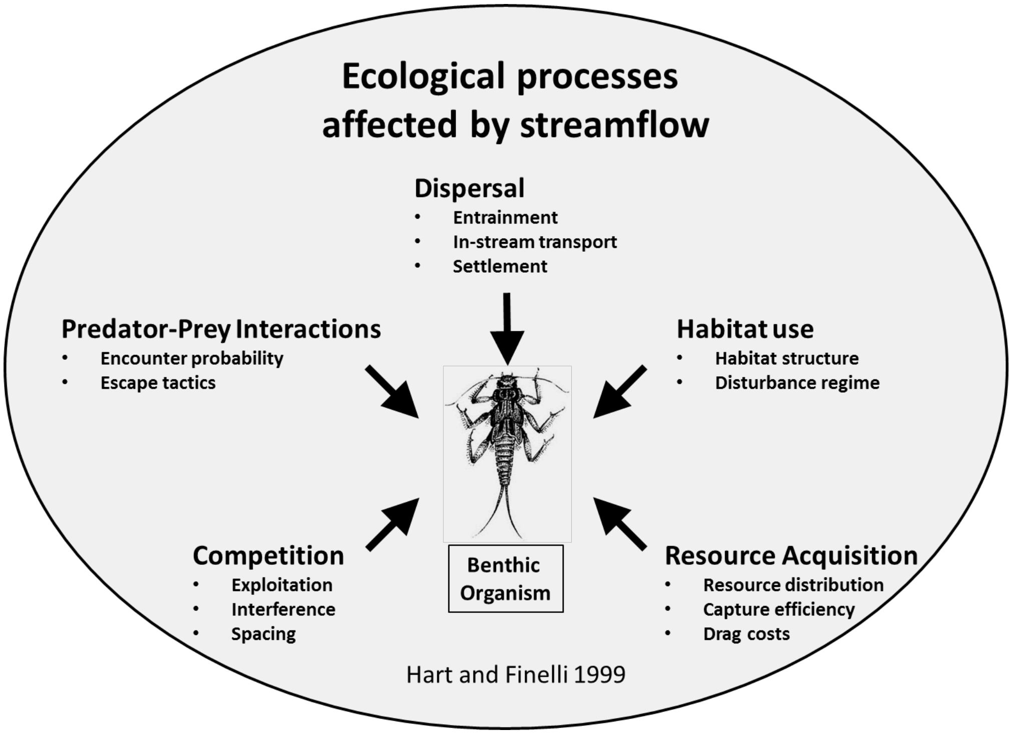


Figure 2.4-2. Substrate habitat suitability indices for benthic invertebrate guilds based upon sampling in the Middle Susitna River during 1984. Source: Hansen and Richards (1985).



Poff et al. 1997

Figure 3.1-1. The natural flow regime concept, adapted from Poff et al. 1997.



Hart and Finelli 1999

Figure 3.1-2. Alternative causal pathways by which flow can affect benthic macroinvertebrates. Key components of each ecological process can be modified by flow, thereby affecting the performance, distribution, and abundance of benthic macroinvertebrates (taken from Hart and Finelli 1999).

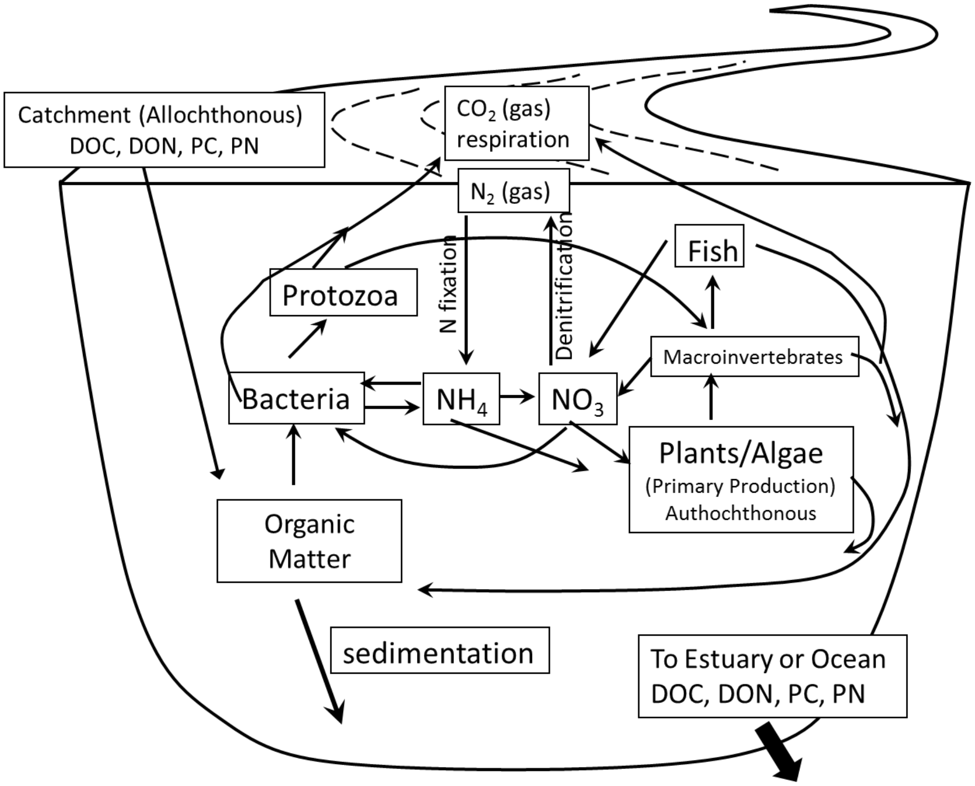


Figure 3.6-1. Linked carbon and nitrogen cycles in river and lake/reservoir ecosystems.

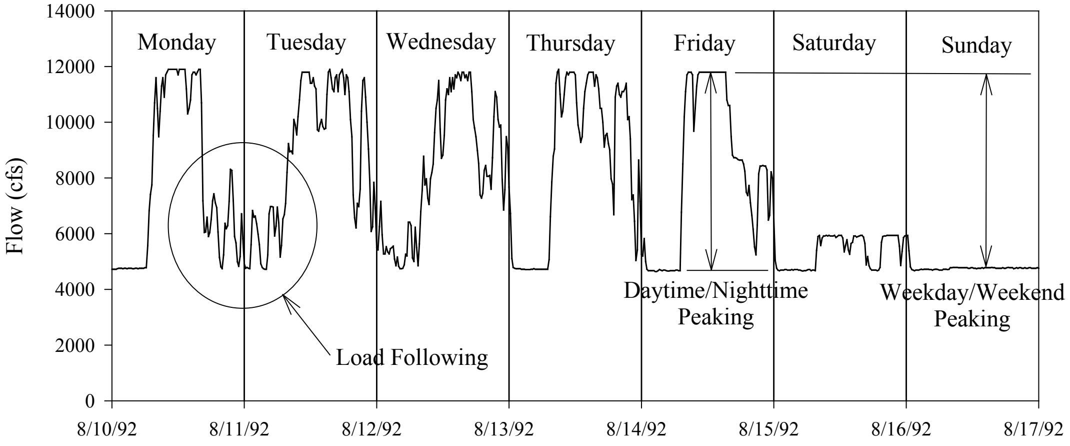


Figure 4.1-1. Example of hydroelectric power peaking and load following operations that result in frequent, large magnitude, short duration pulse type flows. Peaking and load following patterns are primarily evident during the weekdays. Data is from the Flathead River below Kerr Dam, Montana that was historically operated as a peaking/load following facility.

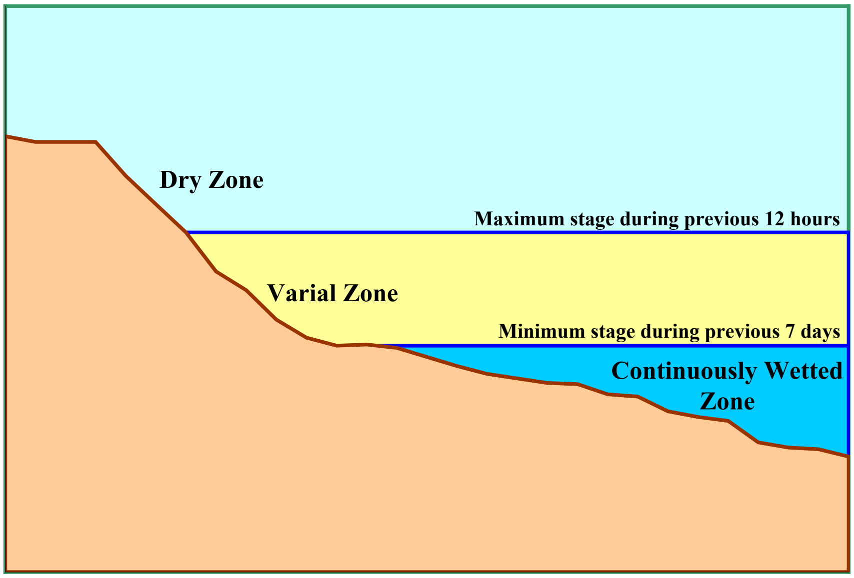


Figure 4.1-2. Example cross-section of a hypothetical channel margin that depicts extent of varial zone as defined by maximum stage of pulse type flow during previous 12 hours. Based on studies conducted on the lower Skagit River, Washington.

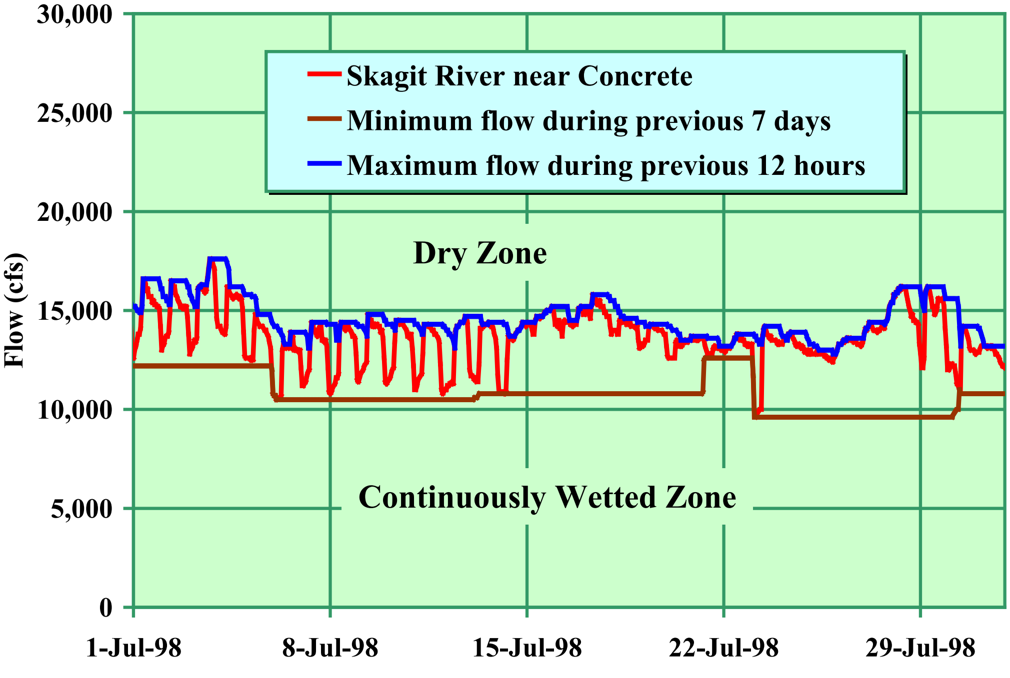


Figure 4.1-3. Example of pulse type flows in the Skagit River, Washington that have occurred in the past (1998) from load following operations of the Baker Hydroelectric Project.

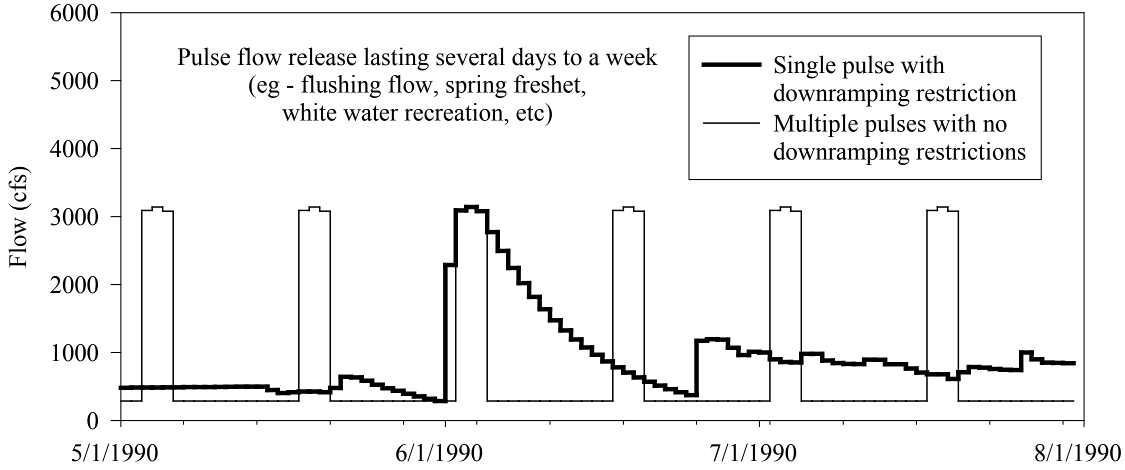


Figure 4.1-4. Examples of infrequent pulse type flows (PTF) that may be associated with flushing flows and recreation flows. Examples depict a PTF "with" and "without" downramping rate restrictions.

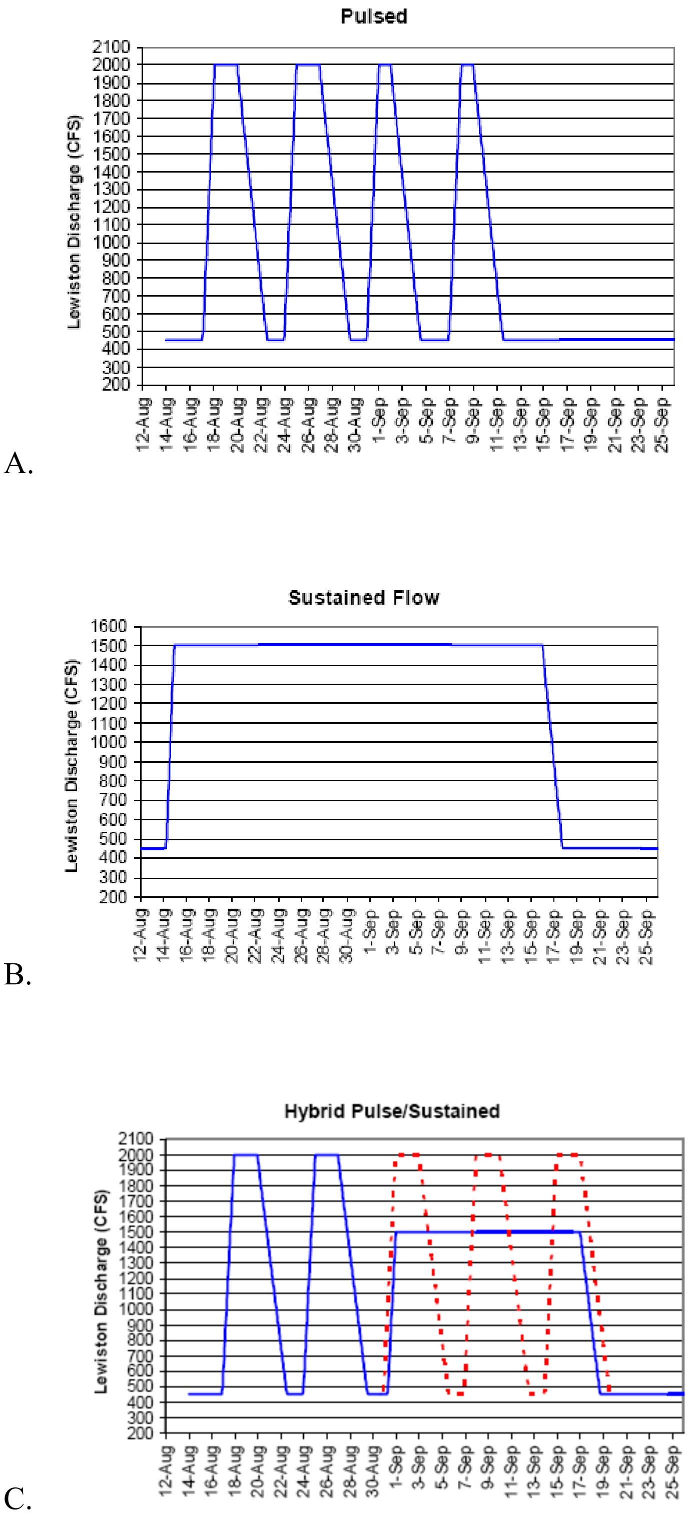


Figure 4.1-5. Various Pulse Type Flows considered for adult salmon attraction for the lower Klamath River, California. A. depicts series of PTF; B. depicts sustained PTF; C. depicts hybrid pulse and sustained PTF. Adapted from Zedonis et al. (2000).

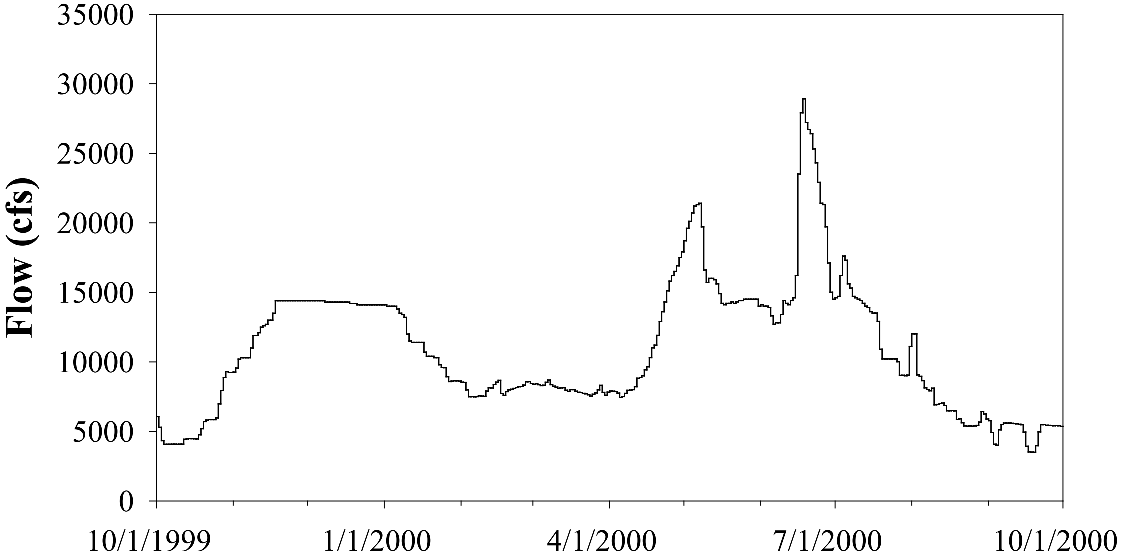


Figure 4.1-6. Example of a baseload operated hydrograph resulting from the operation of the Kerr Dam on the Flathead River, Montana for 1999 and 2000. The shape of the hydrograph is largely determined by resource management objectives that include Flathead Lake management, as well as natural flow conditions. Note that even under baseload operations pulse type flows can occur.

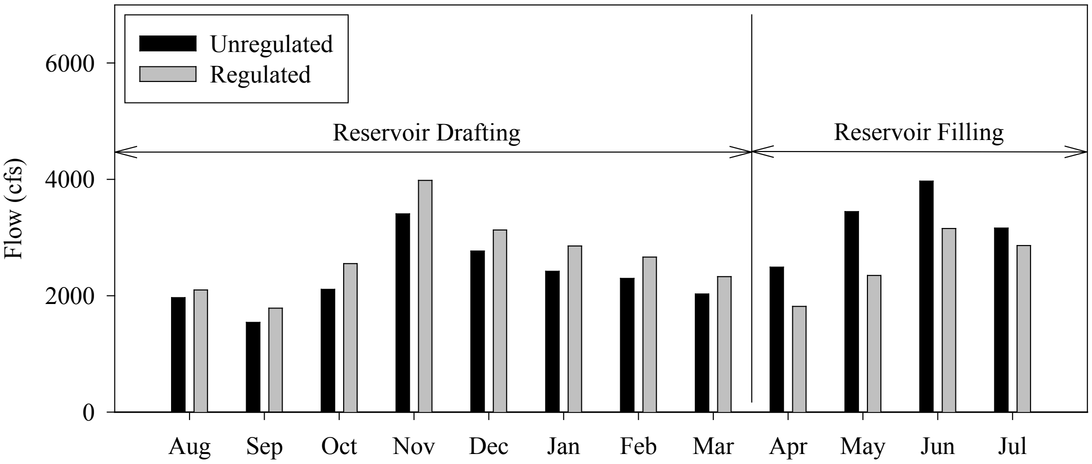


Figure 4.1-7. Comparison of regulated versus unregulated monthly hydrographs for the Flathead River, Montana, below Kerr Dam. Temporal shifts in the occurrence of peak flows results from reservoir drafting and filling.

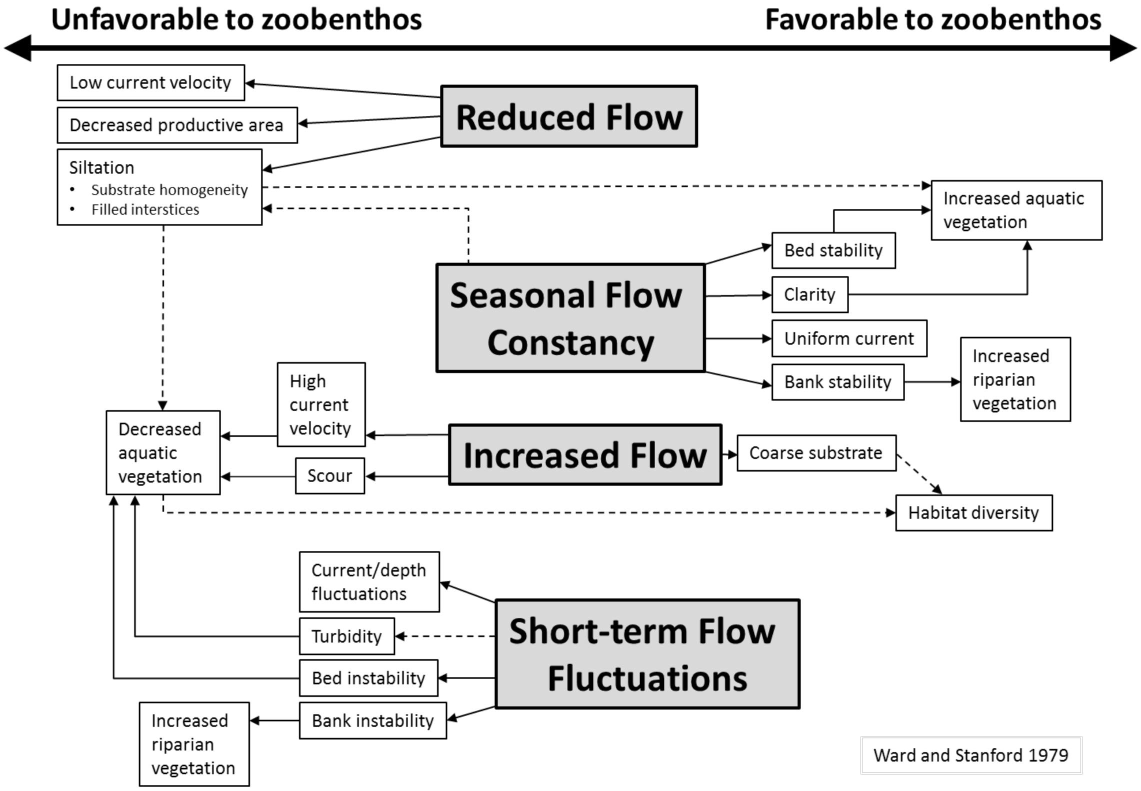


Figure 5.1-1. The potential effects of flow regime modifications below dams on benthic macroinvertebrate communities. Dashed lines represent less definite relationships. (Adapted from Ward and Stanford 1979).

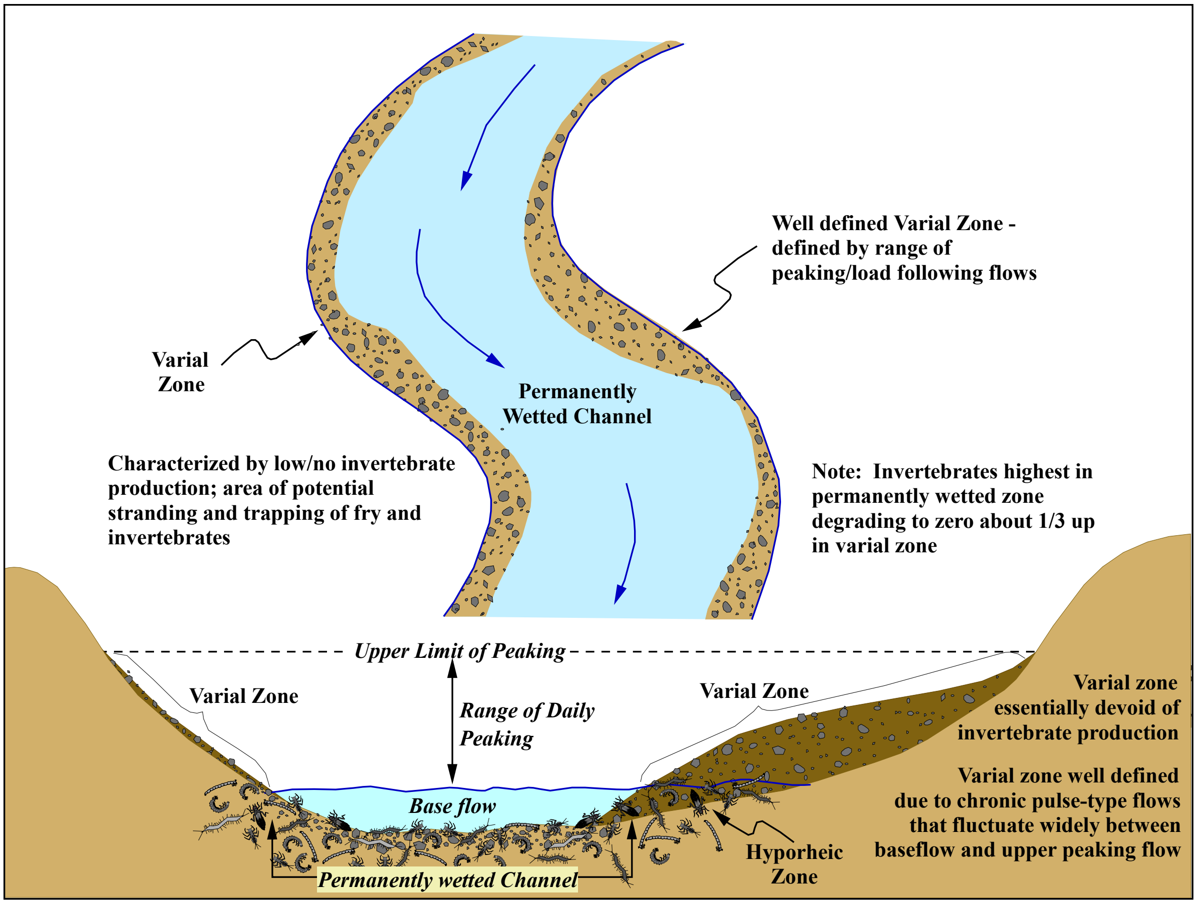


Figure 5.1-2. Schematic planform and cross-section of a stream subjected to daily power peaking pulse type flows illustrating the extent of the varial zone.

Figure 5.1-3. Photograph depicting color gradation of substrate demarcating location of varial zone. The brighter colored rocks are periodically dewatered during the flow reduction phase of the pulse type flow, and therefore lack algae and periphytic growth. (Photo of lower Skagit River, Washington).

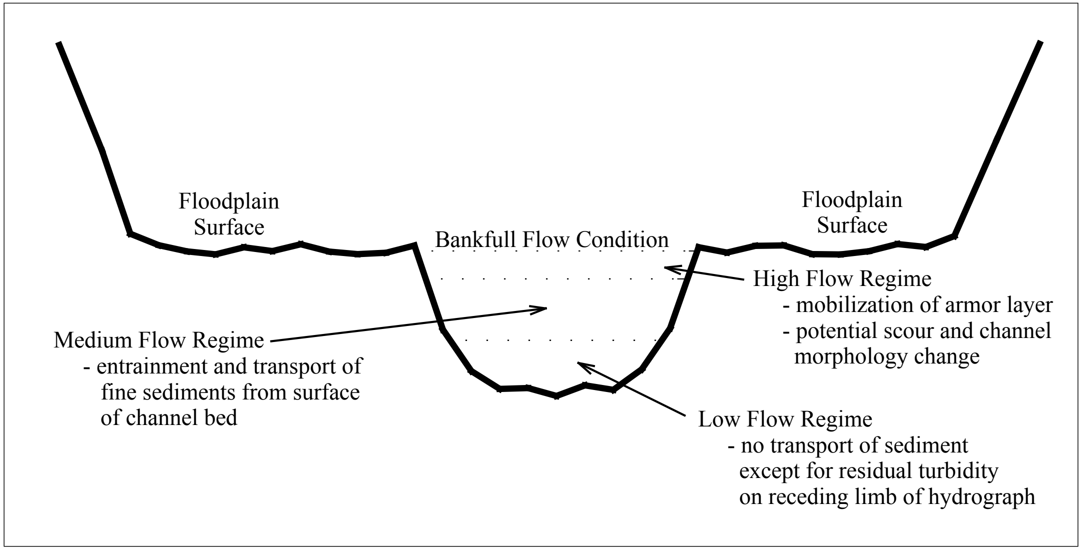


Figure 5.3-1. Conceptual illustration of three flow regimes based on capability of flow to transport sediment and potentially alter the channel morphology.

Susitna-Watana Hydroelectric Project
(FERC No. 14241)

River Productivity Study (9.8)

Appendix B
Site-Specific Sample Collection Locations

Initial Study Report

Prepared for

Alaska Energy Authority

SUSITNA-WATANA HYDRO

Clean, reliable energy for the next 100 years.

Prepared by

R2 Resource Consultants, Inc.

February 2014 Draft

The twenty-three figures in this appendix depict the approximate locations of samples taken at the 20 sites established by the River Productivity Study, as recorded by GPS units at the time of collection during the three index events, as well as those collections related to storm events. Data developed in support of the ISR is available for download at http://gis.suhydro.org/reports/isr (ISR_9_8_RIVPRO_SampleLocations).

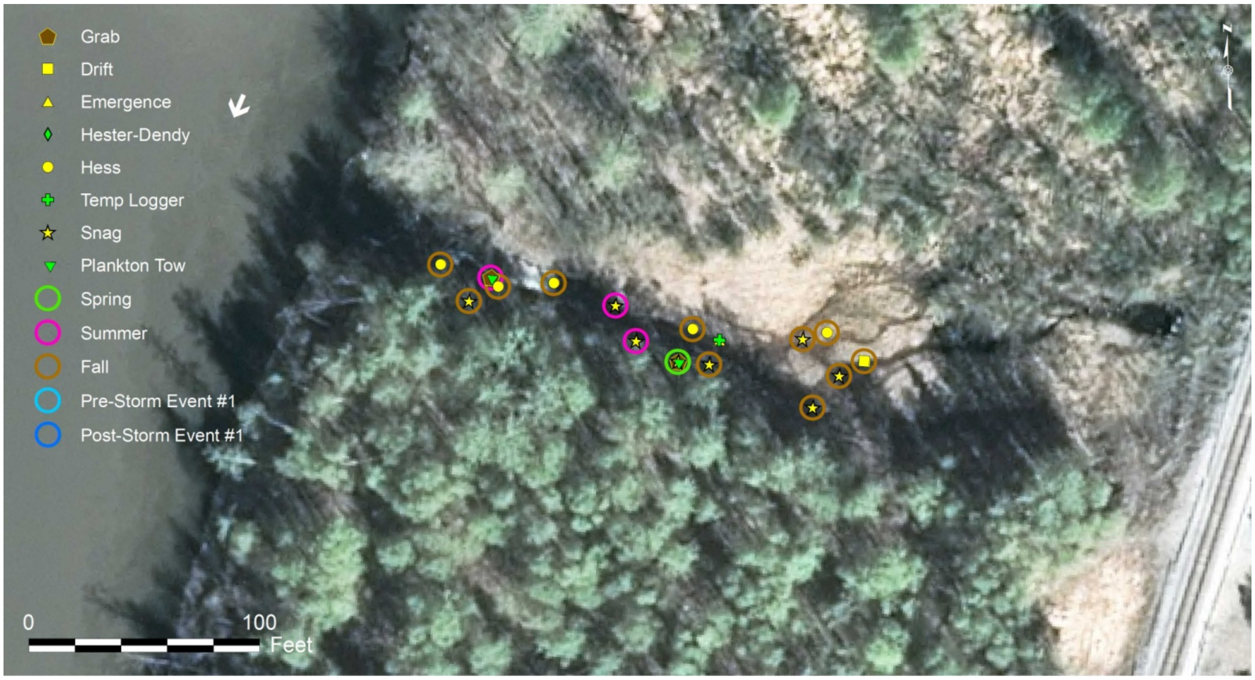


Figure B-1. Locations for samples taken at RP-81-1, an upland slough, near Montana Creek during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

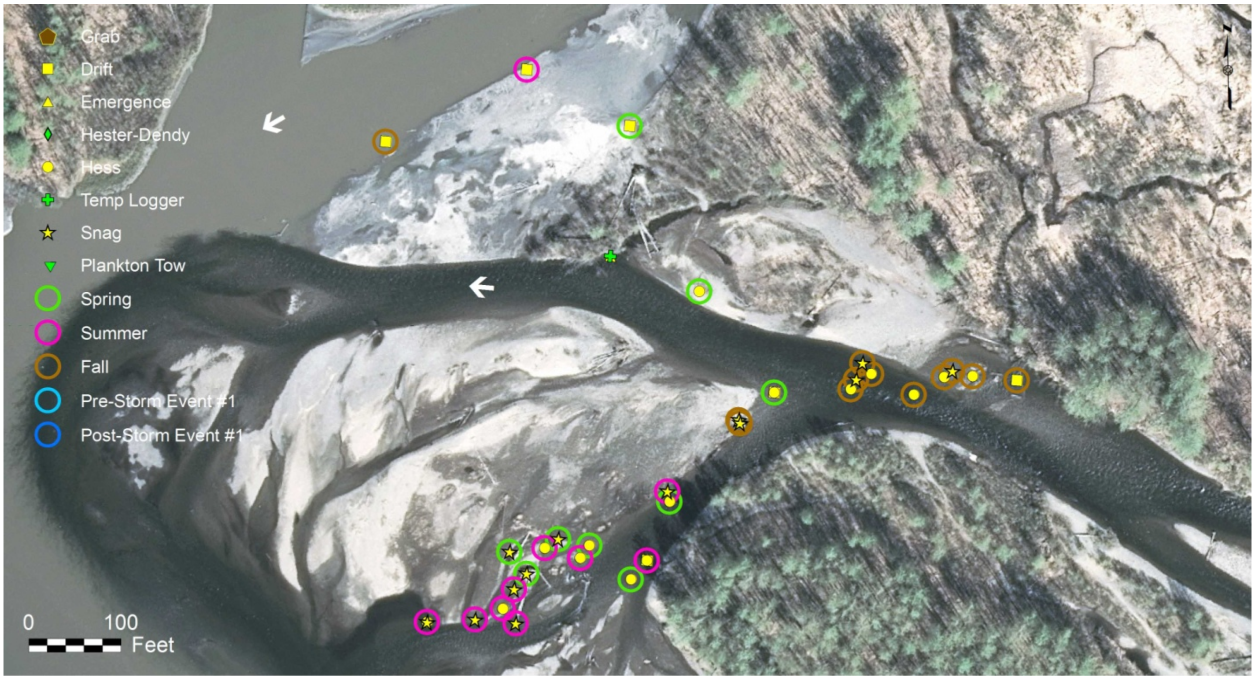


Figure B-2. Locations for samples taken at RP-81-2, the mouth of Montana Creek, during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

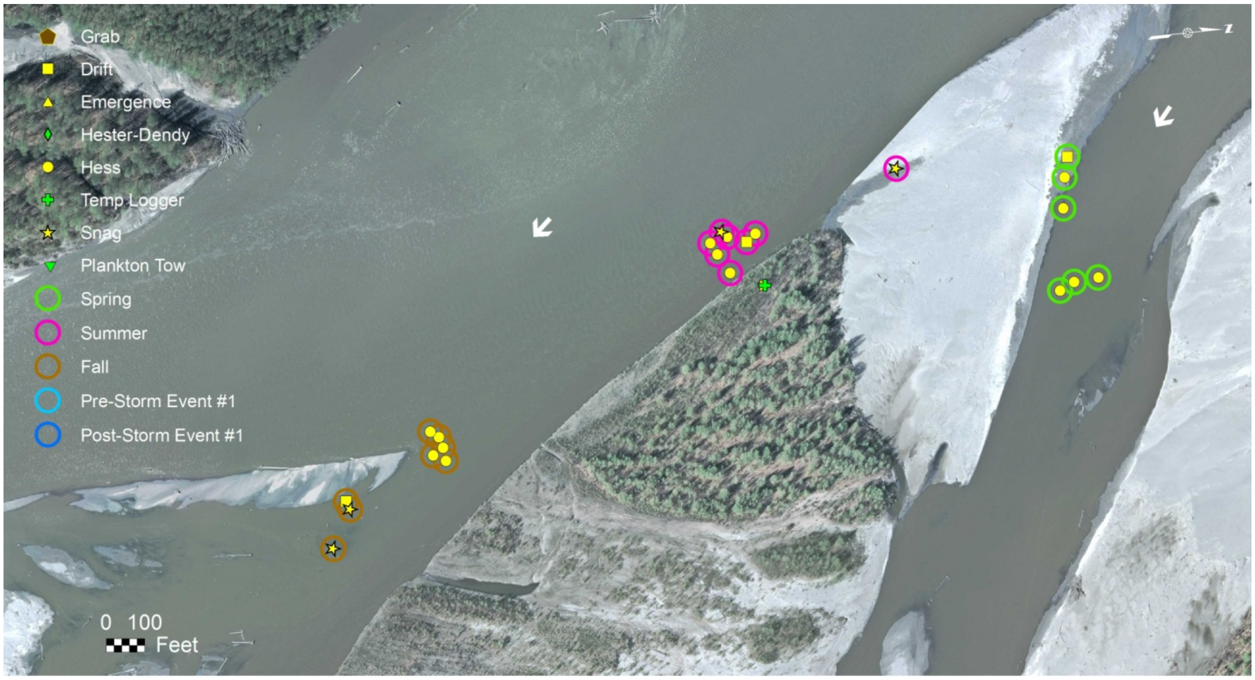


Figure B-3. Locations for samples taken at RP-81-3, a main channel macrohabitat site near Montana Creek, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

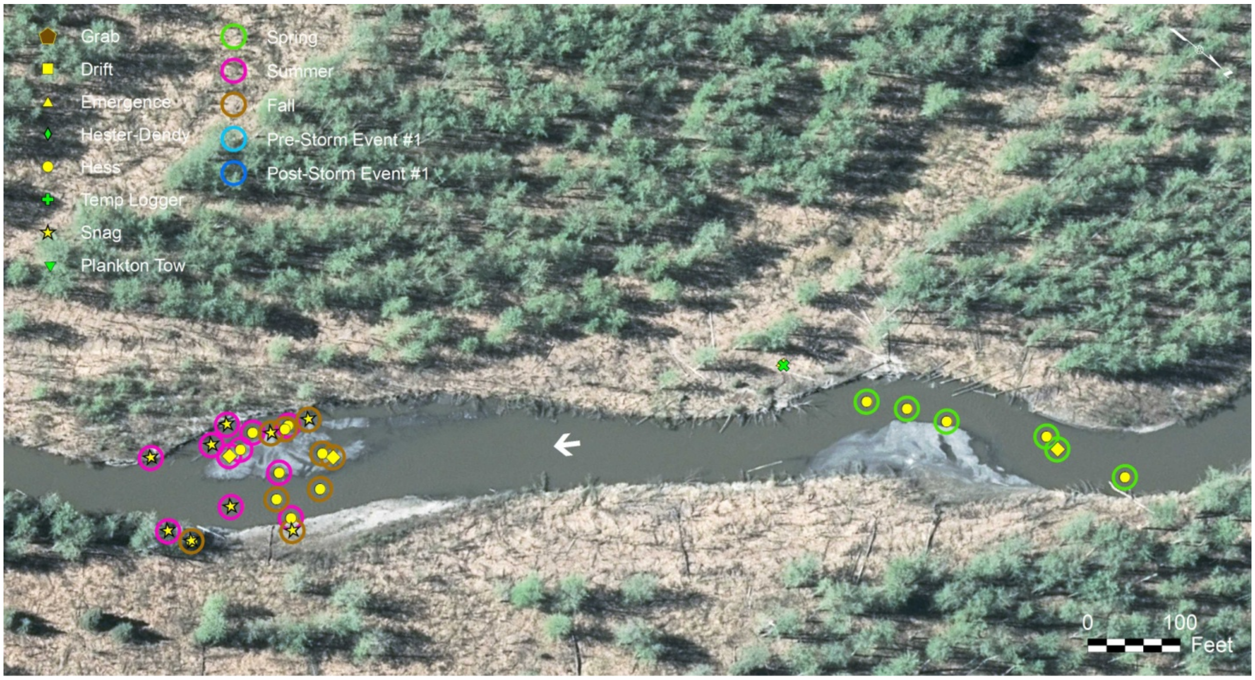


Figure B-4. Locations for samples taken at RP-81-4, a side channel macrohabitat site near Montana Creek, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-5. Locations for samples taken at RP-104-1, a side slough macrohabitat site below the mouth of Whiskers Creek in FA-104, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-6. Locations for samples taken at RP-104-2, a side slough macrohabitat site above the mouth of Whiskers Creek in FA-104, during the three seasonal index events (Spring, Summer/Pre-Storm, Fall) and the Post-Storm Event. Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

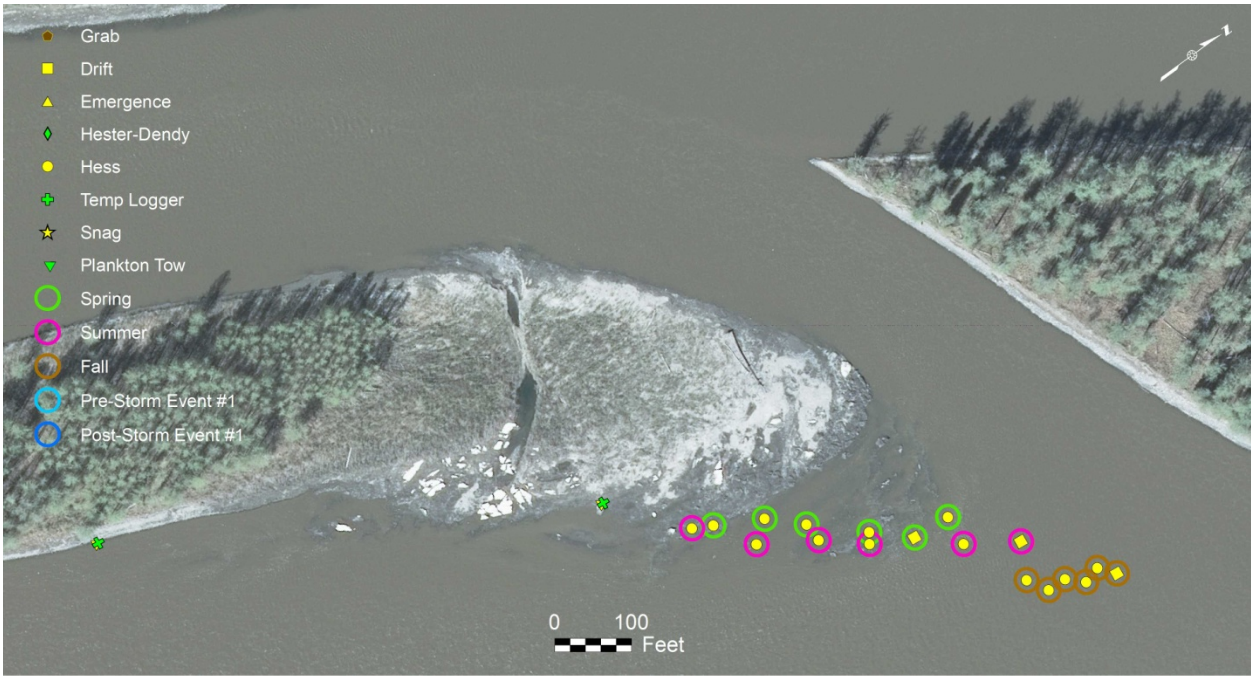


Figure B-7. Locations for samples taken at RP-104-3, a main channel macrohabitat site in FA-104, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-8. Locations for samples taken at RP-104-4, an upland slough macrohabitat site in FA-104, during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

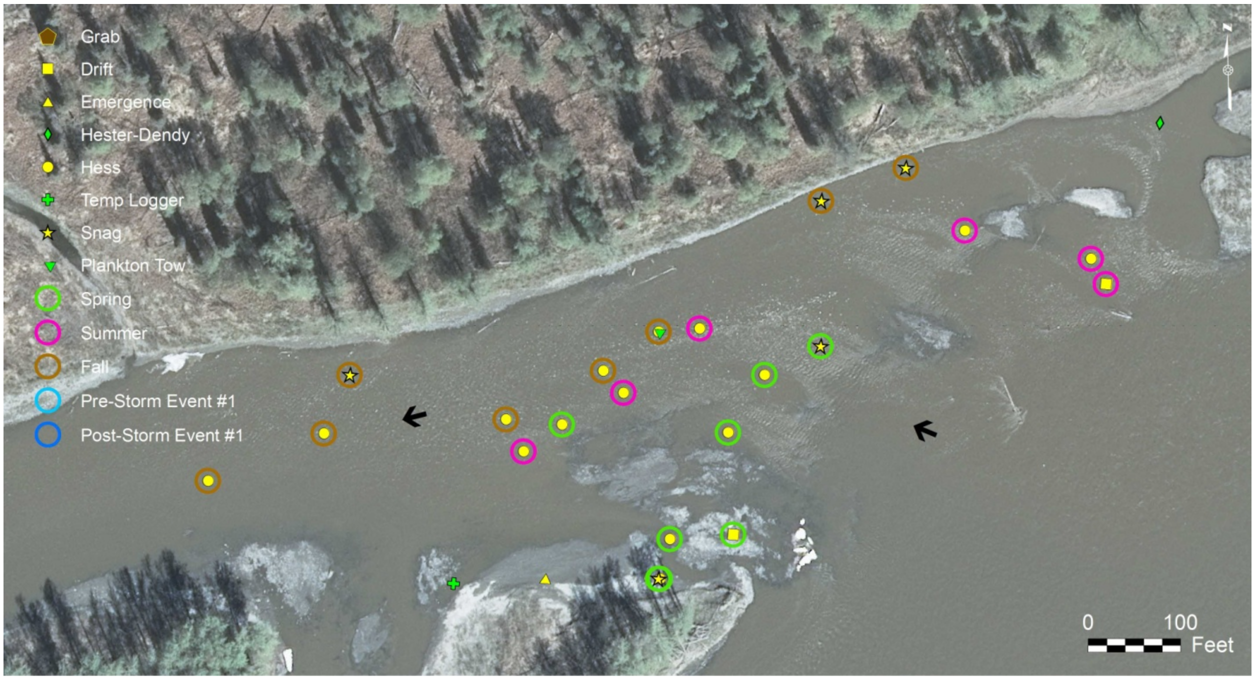


Figure B-9. Locations for samples taken at RP-104-5, a side channel macrohabitat site in FA-104, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-10. Locations for Hester-Dendy samples taken under turbid conditions, located at FA-104. The turbid/cold site, RP-HD-3 (lower left), is located at the upper end of RP-104-5, near the outflow of a small side slough. The turbid/warm site, RP-HD-4 (upper right), is located along the shoreline of a side channel upstream of the upper extent of FA-104.

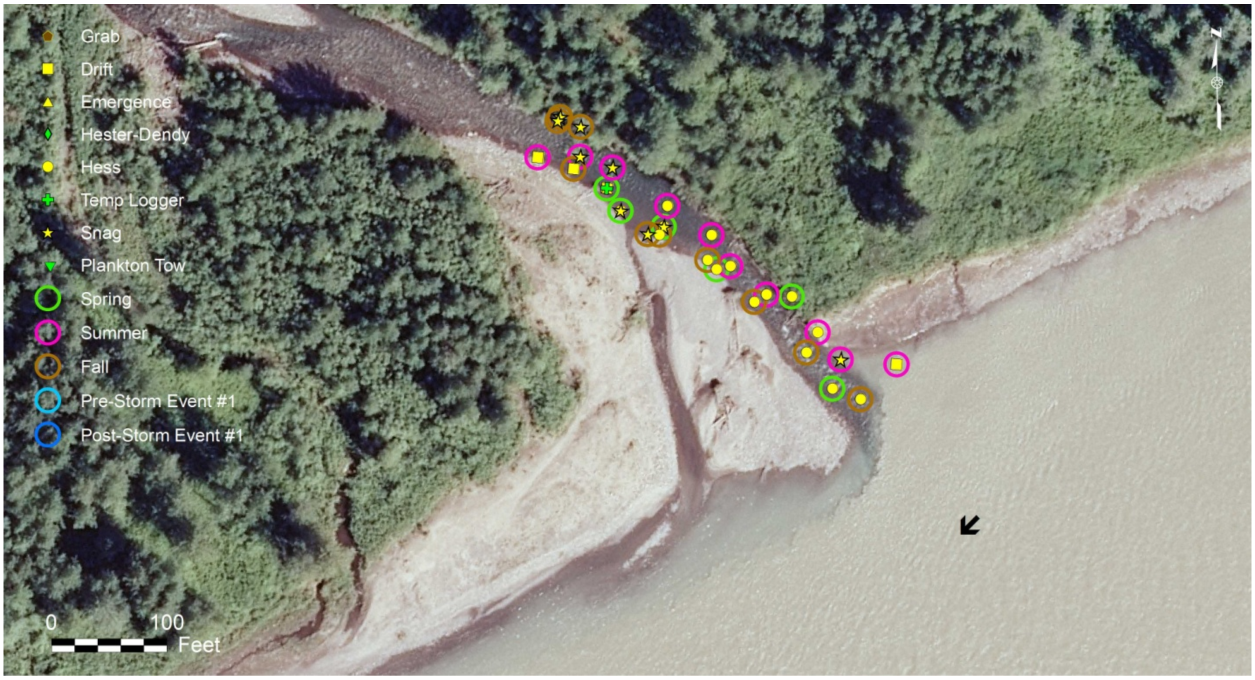


Figure B-11. Locations for samples taken at RP-141-1, a tributary mouth macrohabitat site at Indian River in FA-141, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-12. Locations for samples taken at RP-141-2, a side channel macrohabitat site in FA-141, during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

Figure B-13. Locations for samples taken at RP-141-3, a split main channel macrohabitat site in FA-141, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-14. Locations for samples taken at RP-141-4, an upland slough macrohabitat site in FA-141, during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

Figure B-15. Locations for samples taken at RP-173-1, a tributary mouth macrohabitat site at an unmarked tributary in FA-173, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

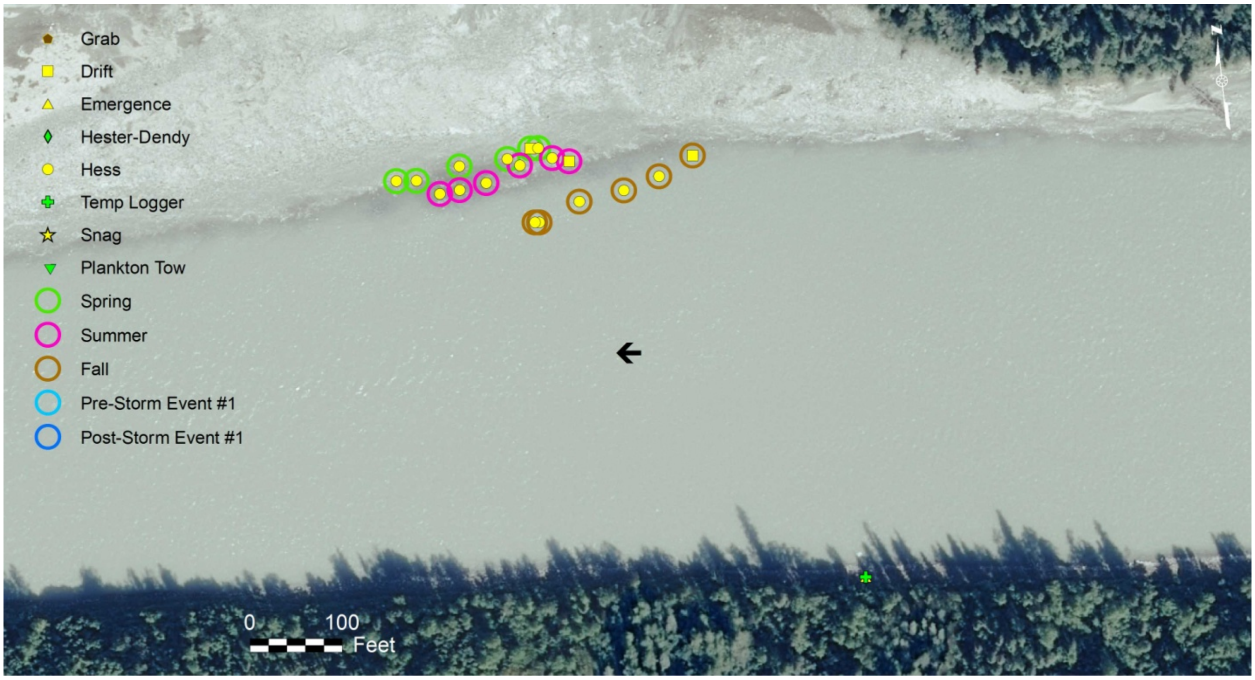


Figure B-16. Locations for samples taken at RP-173-2, a main channel macrohabitat site in FA-173, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

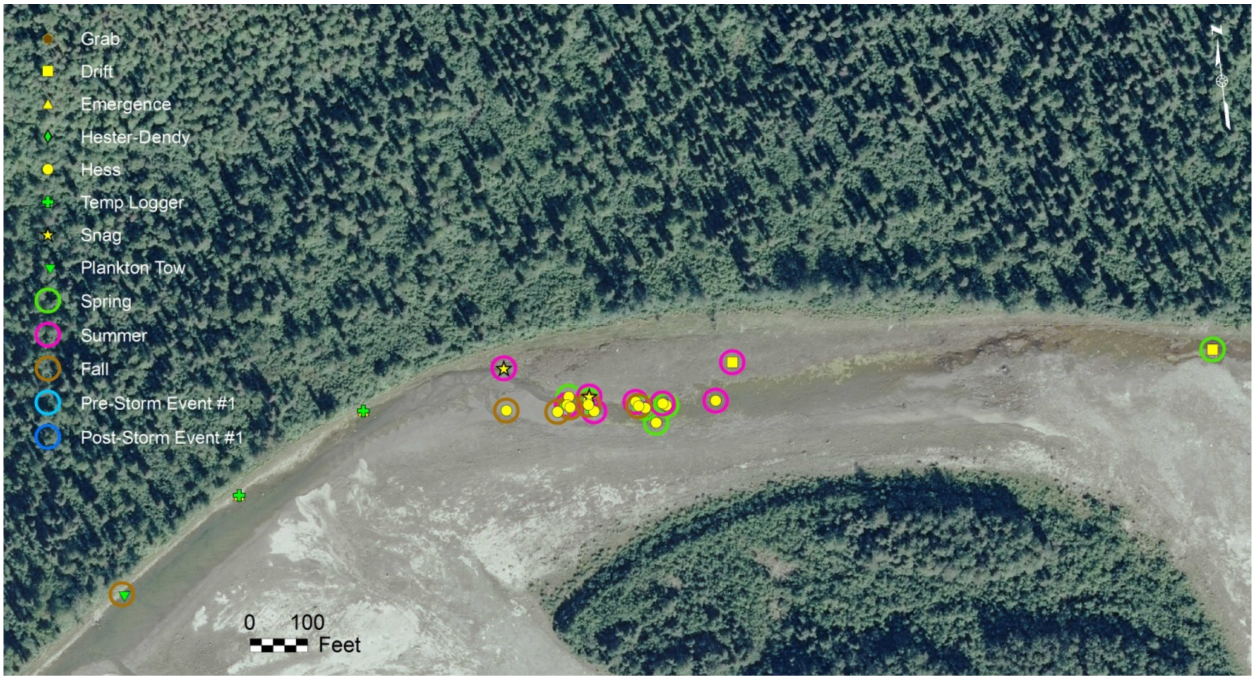


Figure B-17. Locations for samples taken at RP-173-3, a side channel macrohabitat site in FA-173, during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples. Drift markers are representative of duplicate samples.

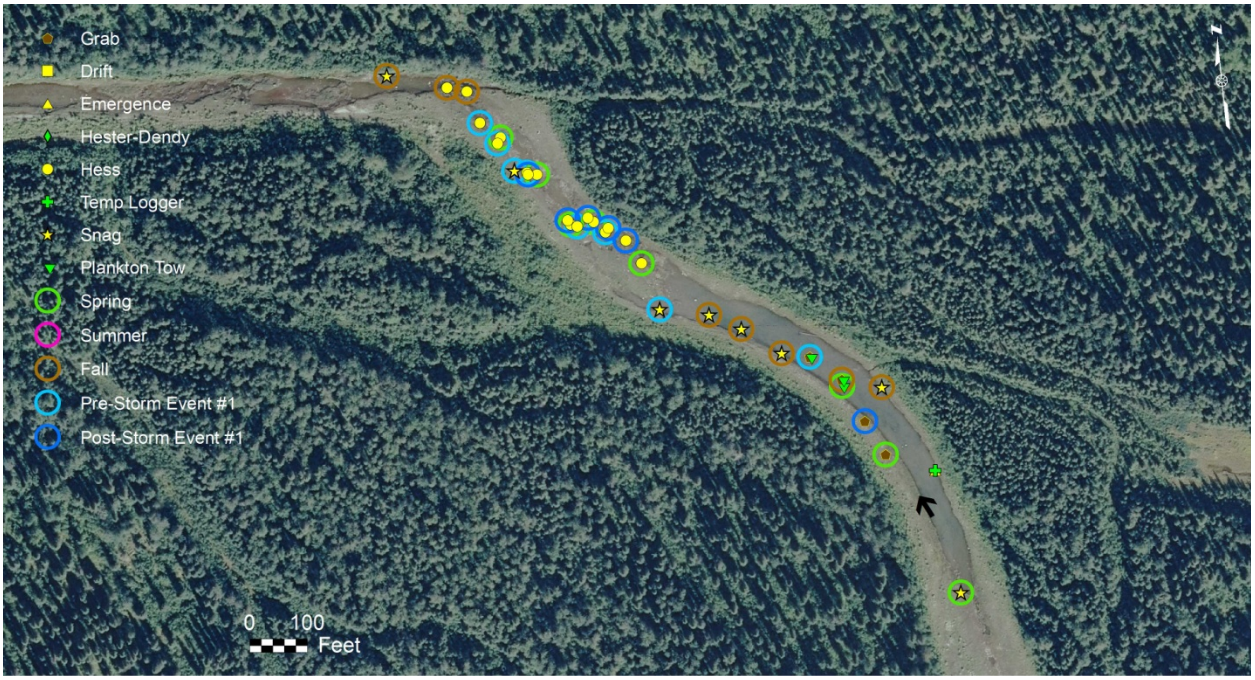


Figure B-18. Locations for samples taken at RP-173-4, a side slough macrohabitat site in FA-173, during the three seasonal index events (Spring, Summer/Pre-Storm, Fall), and the Post-Storm Event. Grab and plankton tow markers are the representative location of 5 replicate samples.

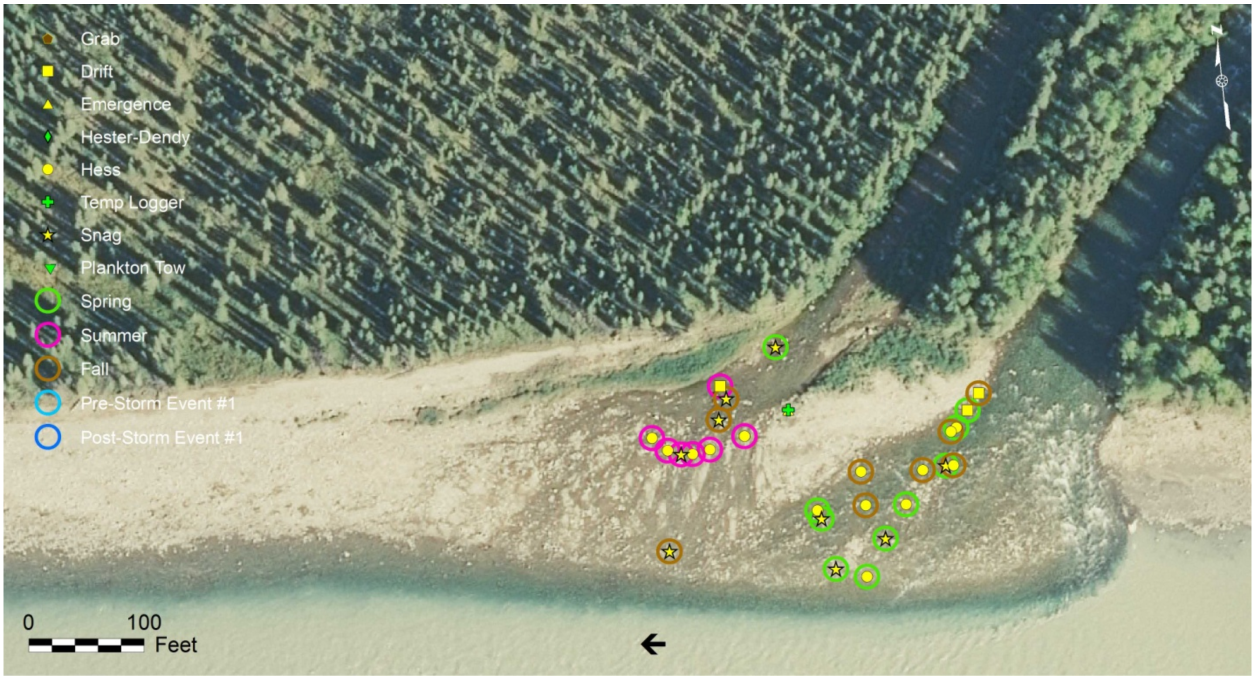


Figure B-19. Locations for samples taken at RP-184-1, a tributary mouth macrohabitat site at Tsusena Creek, just downstream of FA-184, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-20. Locations for samples taken at RP-184-2, a side channel macrohabitat site (top sample group) and RP-184-3, a main channel macrohabitat (bottom sample group), in FA-184 during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-21. Locations for samples taken at RP-TKA-1, a side channel macrohabitat in the Talkeetna River, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Figure B-22. Locations for samples taken at RP-TKA-2, an upland slough macrohabitat in the Talkeetna River, during the three seasonal index events (Spring, Summer, Fall). Grab and plankton tow markers are the representative location of 5 replicate samples.

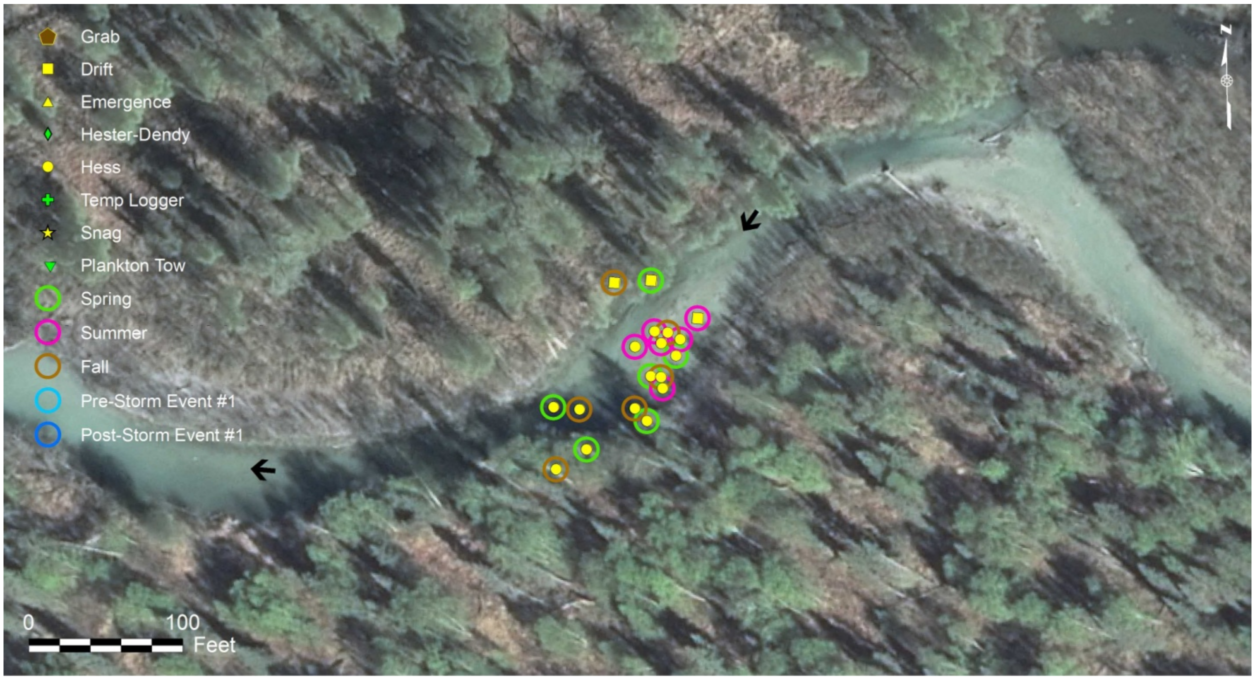


Figure B-23. Locations for samples taken at RP-TKA-3, a side slough macrohabitat site in the Talkeetna River, during the three seasonal index events (Spring, Summer, Fall). Drift markers are representative of duplicate samples.

Susitna-Watana Hydroelectric Project
(FERC No. 14241)

River Productivity Study (9.8)

Appendix C
Analysis of Potentially Dewatered River Productivity Sampling Sites in 2013

Initial Study Report

Prepared for

Alaska Energy Authority

SUSITNA-WATANA HYDRO

Clean, reliable energy for the next 100 years.

Prepared by

R2 Resource Consultants, Inc.

February 2014 Draft

1. INTRODUCTION

On September 23, 2013, a presentation was given to the Technical Working Group (TWG) regarding the third quarter progress of the River Productivity Study. As part of that presentation, variances from the Study Plan were listed, among them the variance regarding sampling site inundation requirements (ISR Section 4.4.3.1). Briefly, the Study Plan stated that: "*Higher flows may inundate new shoreline substrates, which poses the risk of sampling in areas that are not fully colonized. The shoreline bathymetry for each site will be evaluated such that changes in water level due to increasing or decreasing flows must remain constant enough that the substrates accessible for sampling will be continually inundated for a period of at least one month, to facilitate colonization of those substrates*" (RSP Section 9.8.4.3.; IP, Section 2.2.1.). While the preferred and intended practice was to conduct sampling at sites where all substrates had remained submerged for 30 days or more, the constant water levels required to keep sampled areas inundated for at least 30 days were rare; instead the river underwent rapid and sudden changes in flow and river stage during the 2013 sampling season.

Representatives from the Alaska Department of Fish and Game requested that an estimate of the proportion of samples subject to dewatering be provided as part of the ISR. This appendix will detail the approach taken to provide an estimate of how many Hess samples taken during 2013 on the Susitna River were potentially dewatered during the 30 days prior to being collected.

2. METHODS

Flow and river stage data was downloaded from the USGS National Water Information System for stream gage stations located on the Susitna River to determine the change in water levels at each site (Table C-1). Given the significant distance of the sites at FA-104 from the Gold Creek gage (#15292000), a 4 hour 45 minute delay (supplied by the Instream Flow Study team, ISR Study 8.5) was applied to the stage data to account for the transit time from the Gold Creek gage to PRM 104.5. In addition, the stage data for the Sunshine gage (#15292780) was incomplete, with data not recorded until August 4, 2013. Using a rating curve for the Sunshine Station stream gage supplied by the Instream Flow Study team (ISR Study 8.5), the following formula was applied to the flow data at Sunshine station to get an estimated stage at FA-104 to determine potential dewatering.

$$\text{Sunshine Stage} = 9.91 + \frac{Q^{0.6376}}{136.0}$$

A suite of metrics were collected along with each Hess sample obtained during the 2013 field efforts (Table 4.8-1). The date and time of sample collection and the water depth was recorded for each Hess sample. The timing of each Hess sample was then matched to the nearest 15-minute stage measurement at the corresponding stream gage. Each recorded Hess depth was subtracted from the coordinating river stage measurement to provide a "Hess stage". This "Hess stage" represented the river stage required to dewater the sampling location. The "Hess stages" were then further adjusted by 0.1 ft, to reflect the depth into the substrate where each Hess sample was taken. This data developed in support of the ISR is available for download at http://gis.suhydro.org/reports/isr (ISR_9_8_RIVPRO_HessDepthStage).

All 15-minute stage measurements in the previous 30 days were then searched for values that were less than this adjusted "Hess stage". Those counts were multiplied by 15 to estimate the total number of minutes that the sampling site was potentially dewatered, along with the date and time of the most recent potential dewatering event. These results were compiled in a summary table for each site (Table C-2).

3. RESULTS

Results from this analysis initially revealed that 137 out of 271 Hess samples, roughly 50-percent, may have been collected in areas that were potentially dewatered in the prior 30 days, with most occurring during the spring and summer index event (Table C-2). However, this result should be interpreted with caution. The use of USGS stream gage stage data for this analysis required the following assumptions as to the applicability of the data for site-specific analysis:

1) The analysis assumed that the recorded stage at specific times corresponded to the sample collection times, and that travel time adjustments were correct and applicable.
2) The analysis assumed that changes in Susitna River stage were similar at corresponding main channel sites and areas.
3) The analysis assumed that the river stage levels from stream gages could account for the influence of local hydraulics. Changes in the river stage were likely less applicable at off-channel sites and habitats (tributary mouths, side sloughs, upland sloughs, and some side channels) compared to main channel macrohabitat sites (main channels, side channels).

Given these assumptions, samples collected in off-channel macrohabitats would have been less influenced by Susitna River stage changes than main channel macrohabitats. Tributary mouths generally would be continually watered by stream flow from the tributary itself. Upland sloughs and side sloughs, once disconnected from the main channel flows, would have likely remained inundated in a series of pools fed by groundwater, or upwelling intergravel flow. Of the 137 Hess samples potentially collected in previously dewatered areas, 72 were located in off-channel macrohabitats, and 65 were in main channel macrohabitats.

Considering only main channel macrohabitat sites, approximately 24-percent of the Hess samples collected in 2013 were potentially dewatered in the 30 days prior to sampling. Further examination by index event showed that 34-36 percent of samples collected in spring and summer events were at risk of having been previously dewatered, whereas the fall index event included only 2 potentially dewatered samples, both at RP-104-5, the side channel site in FA-104. Examination of the river stage over the 2013 open-water season shows high variability in the stage levels at each of the sampling stations during the spring and summer index events (Figures C-1 through C-5).

Depths (Hess stages) of all Hess samples collected during 2013 at each of the five sampling stations were plotted with the river stage data for the open-water season (Figures C-1 through C-5). These plots illustrate the periods of lower river stage prior to the collection of Hess samples. For instance, samples collected during the spring index event at FA-104 were taken during a peak in river stage and the declining limb of that increase (Figure C-2). Samples taken at the peak would have in areas dewatered only 5 days earlier. However, most samples taken at RP-104 were inundated for at least 25 days, when spring flows began to increase from winter base flows. In contrast, samples collected on an increasing limb of stage levels would be in newly inundated shoreline areas, as was seen at FA-173 (Figure C-4) and especially at FA-184 (Figure C-5) during the late August storm event. Some samples collected within FA-173 on August 19-20, 2013 were taken at sites that may have been dewatered several times for short periods over the previous 30 days (Figure C-4). Samples collected within FA-184 on August 20-21, 2013 were taken at sites that were most definitely dewatered for the majority of the previous 30 days (Figure C-5). These plots also show that samples collected during the fall index event were collected during a declining river stage, and would have been inundated for the entire 30 day period prior to collection (Figures C-1 through C-4).

4. DISCUSSION AND CONCLUSIONS

While this analysis makes a number of assumptions in order to apply the available river stage data collected by USGS gaging stations on the Susitna River, the exercise does demonstrate the difficulties in adhering to a sampling criterion of 30 days of inundation at a site on the Susitna River. The dynamics of a glacially-fed river system make flow difficult to predict, and the timing of sampling trips are often determined weeks in advance to meet field logistical demands. It is important to consider that benthic macroinvertebrate communities exist under these naturally fluctuating flows and water levels. Sampling under these conditions accurately reflects the benthic community structure as shaped by the seasonally variable flow regime.

Sampling efforts can be adjusted to accommodate for stage changes and to attempt to minimize their effects in 2014. For instance, Hess samples can be taken at the maximum depths available at sites (up to the 14-16 inch height of the sampler unit) to minimize the effects of stage changes. Sampling efforts may also be shifted to sites in sloughs and tributary mouths where the risk of dewartering is reduced due to sources that are independent of mainstem flows. Finally, sampling efforts at the main channel and side channel sites could be rescheduled to later in the open water season when flows and river stages decline.

Table C-1. List of USGS stream gaging stations and the corresponding 2013 River Productivity Stations used in the analysis of potential dewatering in sampled areas.

USGS Station	Station Name	Corresponding River Productivity Stations
1591700	Susitna River Above Tsusena Creek near Chulitna, AK	RP-184 (Watana Dam Site) RP-173 (Stephan Lake Complex)
15292000	Susitna River at Gold Creek, AK	RP-141 (Indian River) RP-104 (Whiskers Slough)
15292780	Susitna River at Sunshine, AK	RP-81 (Montana Creek)

Table C-2. Summary of the potential number of Hess samples collected in previously dewatered habitat areas at River Productivity sites on the Susitna River during the three index events in 2013.

Site	Macro-habitat Type	Potentially Dewatered Hess Samples					
		Spring	**Summer**	**Fall**	**Post-Storm**	**Total Potentially Dewatered**	**Total Collected**
RP-81-1	US					0	5
RP-81-2	TM	5	5	1		11	15
RP-81-3	Split MC	4	4			8	15
RP-81-4	SC	5	5			10	15
RP-104-1	SS	5	3			8	15
RP-104-2	SS	5	5		5	15	17
RP-104-3	MC	5				5	15
RP-104-4	US					0	0
RP-104-5	SC	5	1	2		8	15
RP-141-1	TM	2				2	15
RP-141-2	SC	2	2			4	10
RP-141-3	Mult Split MC		2			2	15
RP-141-4	US	5	1			6	12
RP-173-1	TM	5	5			10	15
RP-173-2	MC	5	3			8	15
RP-173-3	SC	5	5			10	15
RP-173-4	SS	5	5		5	15	17
RP-184-1	TM		5			5	15
RP-184-2	SC		5			5	15
RP-184-3	MC		5			5	15
Totals:		**63**	**61**	**3**	**10**	**137**	**271**
Main Channel Type		31	32	2	0	65	
Off-Channel Type		32	29	1	10	72	

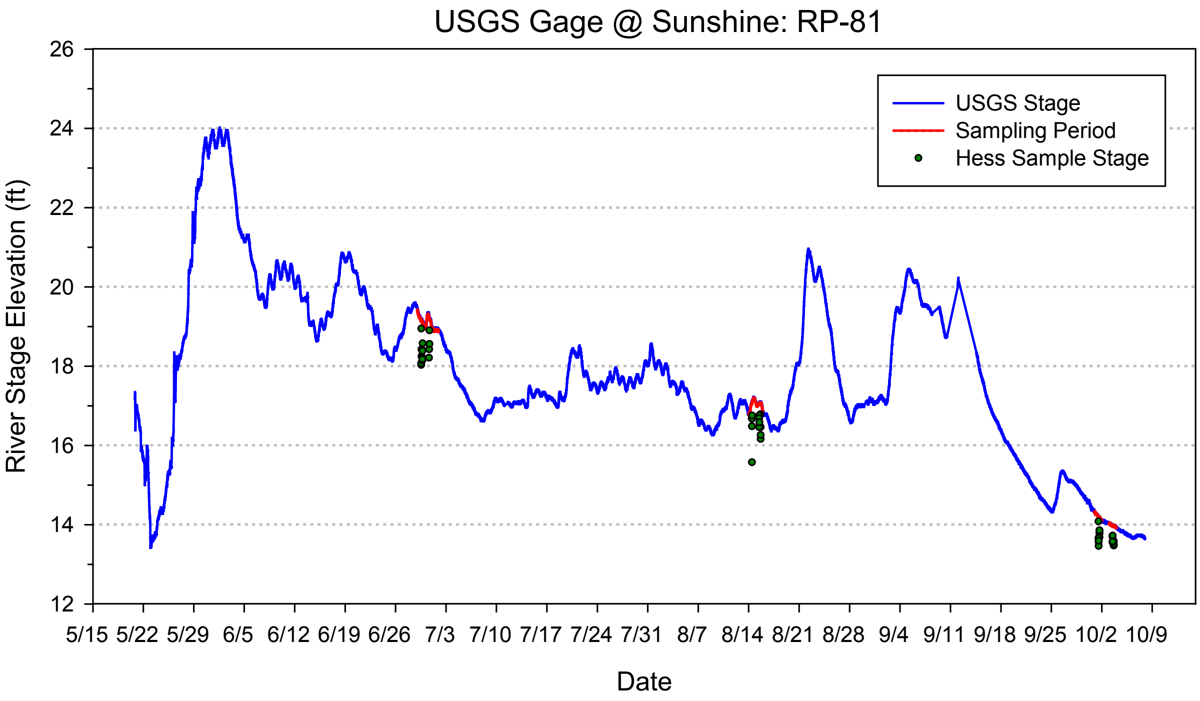


Figure C-1. River stage at the USGS Gage 15292780: Sunshine, AK for May 20 through October 7, 2013, with estimated Hess sample depths at time of collections at River Productivity station RP-81 (Montana Creek) during the 2013 sampling season.

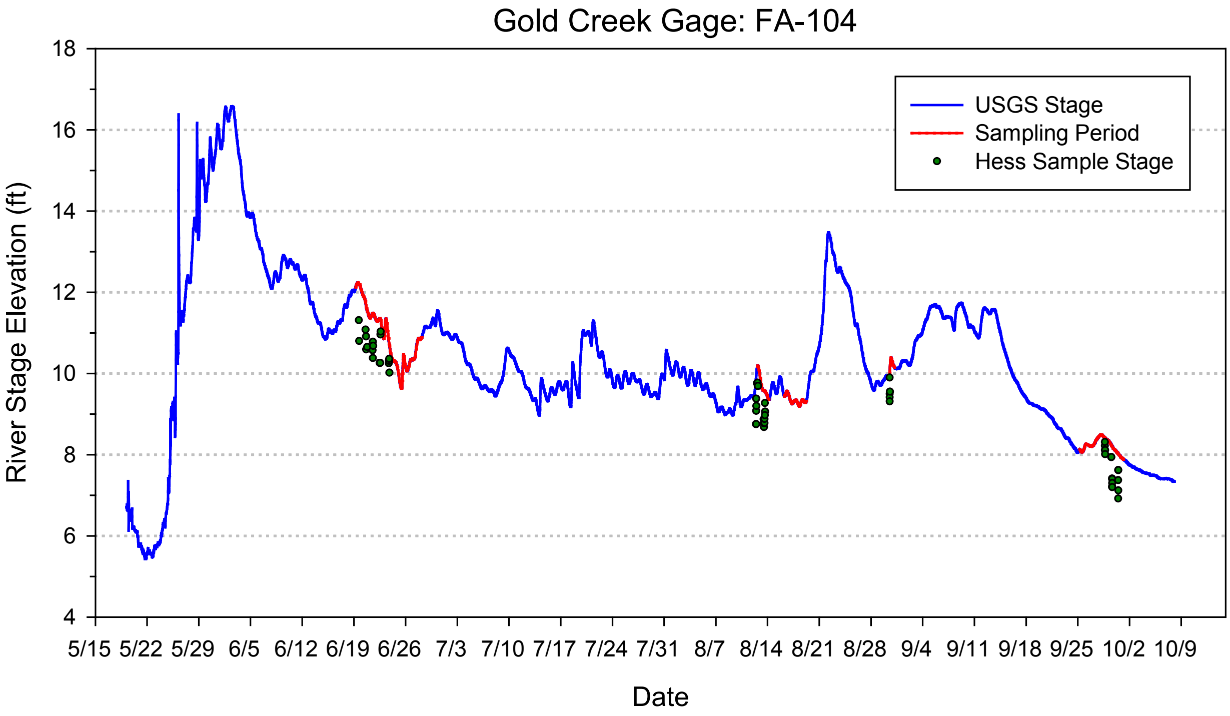


Figure C-2. River stage at the USGS Gage 15292000: Gold Creek, AK for May 19 through October 7, 2013, (adjusted by 4:45 hrs) with estimated Hess sample depths at time of collections at River Productivity station RP-104 (Whiskers Slough) during the 2013 sampling season.

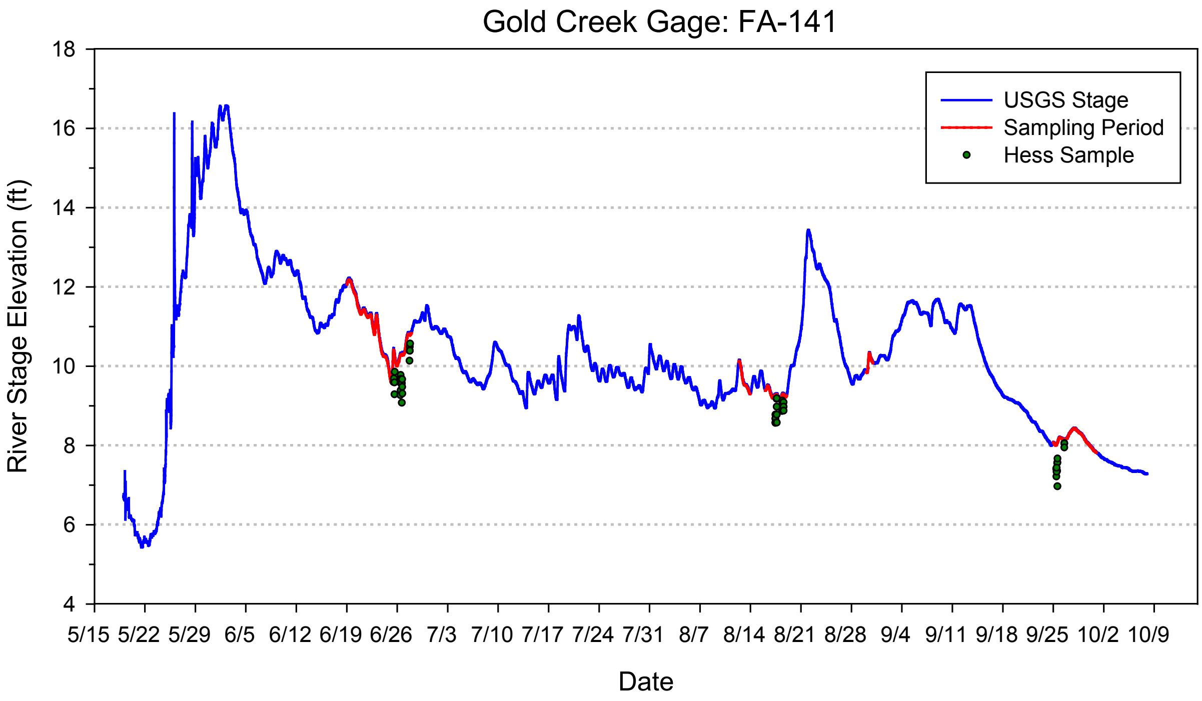


Figure C-3. River stage at the USGS Gage 15292000: Gold Creek, AK for May 19 through October 7, 2013, with estimated Hess sample depths at time of collections at River Productivity station RP-141 (Indian River) during the 2013 sampling season.

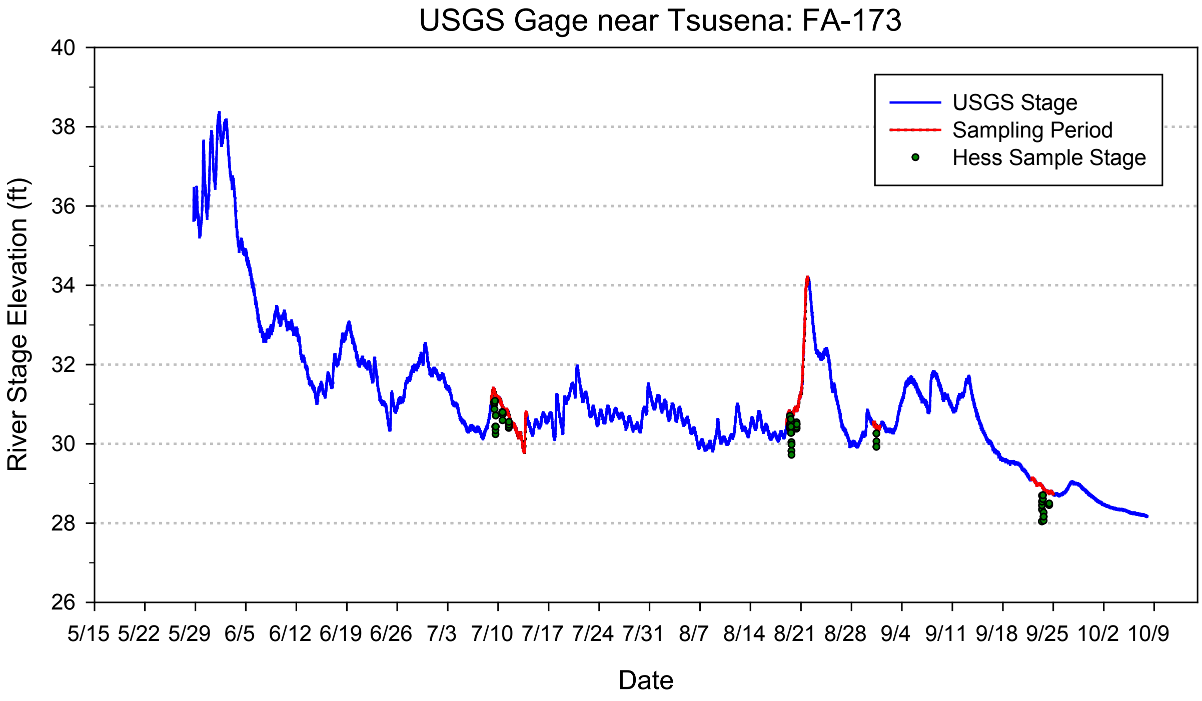


Figure C-4. River stage at the USGS Gage 152921700: near Tsusena Creek for May 28 through October 7, 2013, with estimated Hess sample depths at time of collections at River Productivity station RP-173 (Stephan Lake complex) during the 2013 sampling season.

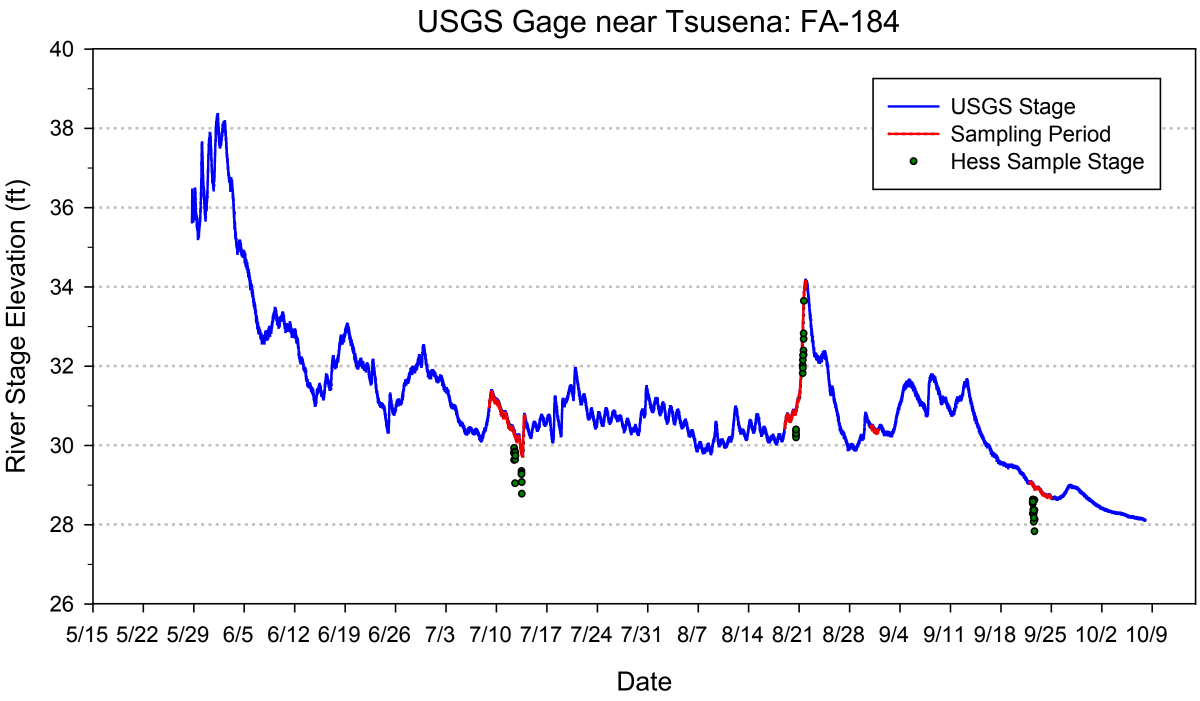


Figure C-5. River stage at the USGS Gage 152921700: near Tsusena Creek for May 28 through October 7, 2013, with estimated Hess sample depths at time of collections at River Productivity station RP-184 (Watana Dam site) during the 2013 sampling season.

Susitna-Watana Hydroelectric Project
(FERC No. 14241)

River Productivity Study (9.8)

Appendix D
Talkeetna Site Selection Consultation Documentation

Initial Study Report

Prepared for

Alaska Energy Authority

SUSITNA-WATANA HYDRO

Clean, reliable energy for the next 100 years.

Prepared by

R2 Resource Consultants, Inc.

February 2014 Draft

EMAIL RECORD

From: Tim L. Nightengale [tnightengale@r2usa.com]
Sent: Friday, July 12, 2013 12:59 PM
To: 'Jeffrey Davis'
Cc: 'Betsy McGregor'; 'marie.steele@alaska.gov'; 'susan walker'; 'Catherine_Berg@fws.gov'; 'michael_buntjer@fws.gov'; 'MaryLou Keefe'
Subject: RE: River Productivity Consult with Agencies on Talkeetna Reference sites
Attachments: Talkeetna site reconn.pptx

Hi Jeff,

As part of the River Productivity Study, one of the objectives (9.8.4.6.) was to “conduct a feasibility study in 2013 to evaluate the suitability of using reference sites on the Talkeetna River to monitor long-term Project-related change in benthic productivity.” AEA consulted with the TWG during the meeting on June 24, 2013 on the Talkeetna River reference sites. Based upon that consultation, I selected some options for possible reference sites on the Talkeetna River for sampling under the RSP task for the River Productivity Study. I was able to set up a quick reconn trip at the end of a full day on June 29 with a boat captain who lives in Talkeetna, and knew the river well. We spent a couple of hours that evening touring the river from the bridge up to the power lines above Clear Creek, visiting Wiggle Creek, Clear Creek, and Fish Creek, and viewing several of the areas along the way.

From the June 29 tour, I was able to better select candidate sites. Upon my return to the office, I was able to access better map imagery, and I have constructed a Powerpoint file that highlights these possible selections (attached). I have circled several candidate sites, as you suggested, and have included some notes/comments.

Also discussed in the TWG meeting on June 24th , AEA has reviewed how many sites are recommended for this particular task by FERC under the RSP, and Implementation Plan. The RSP states “sampling sites will be established in the Talkeetna River in areas that are physically similar to those sampled in the Middle Susitna River Segment, to ensure comparability. Sampling will be conducted in riffle habitats within the mainstem, side channels, and sloughs. One station will be established, with a mainstem site and two off-channel habitat sites associated with the mainstem site.” In addition, the Implementation Plan states “The Talkeetna Station will feature both main channel and off-channel habitat types to allow for the establishment of a main channel site, a side channel site, and a side slough site.” Based upon this, the Talkeetna station for river productivity will include three sites.

I would appreciate it if you could review these selections included in the Powerpoint slides and get back to me with your thoughts on them. Currently, the RivPro field crew is finishing their sampling effort at FA-184 (Watana Dam site), and will be ready and available to start sampling efforts on the Talkeetna River, starting Monday. If your input is provided today or this weekend, AEA will be able to consider this additional input.

Best Regards,
Tim

Tim Nightengale
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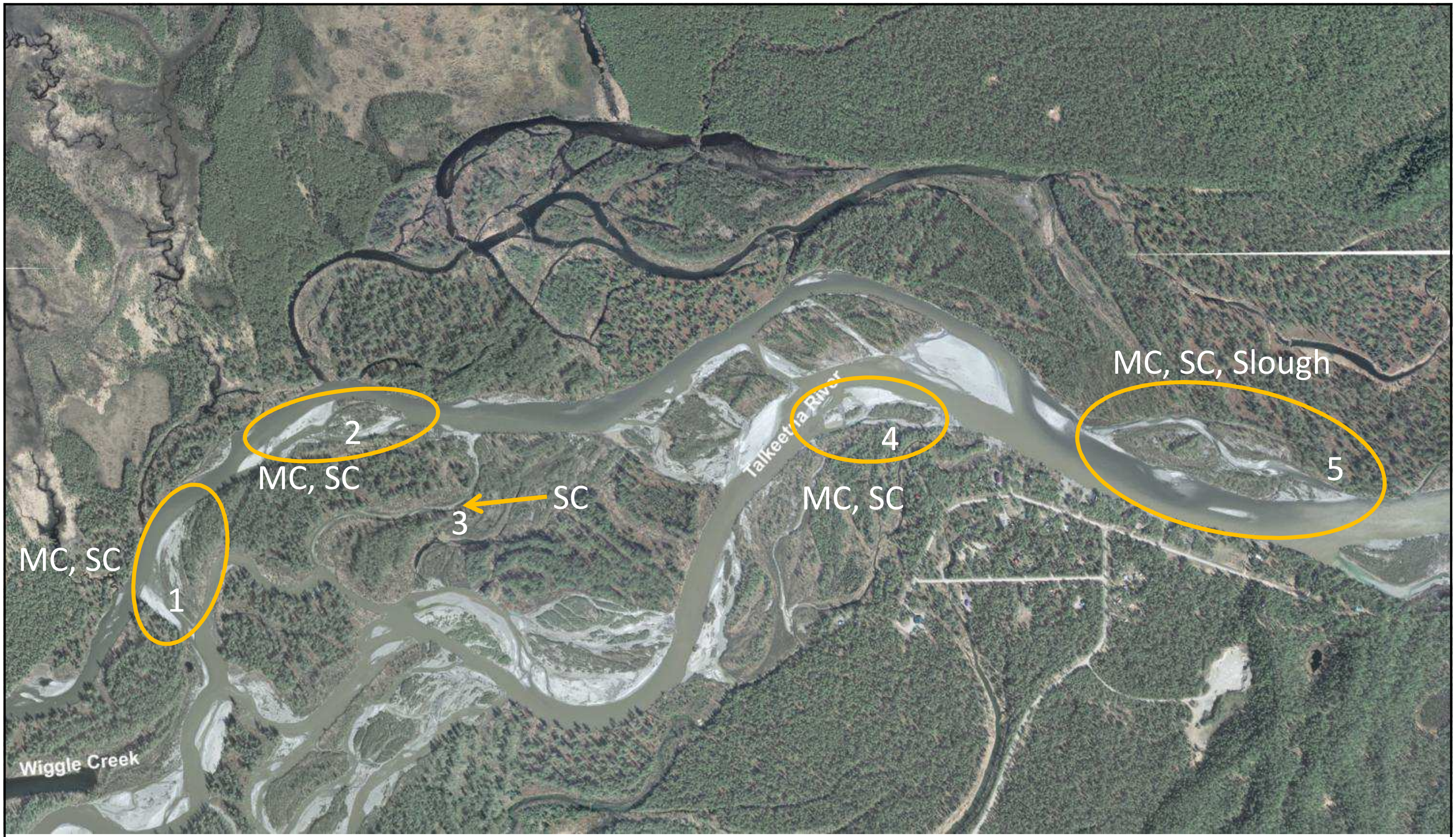
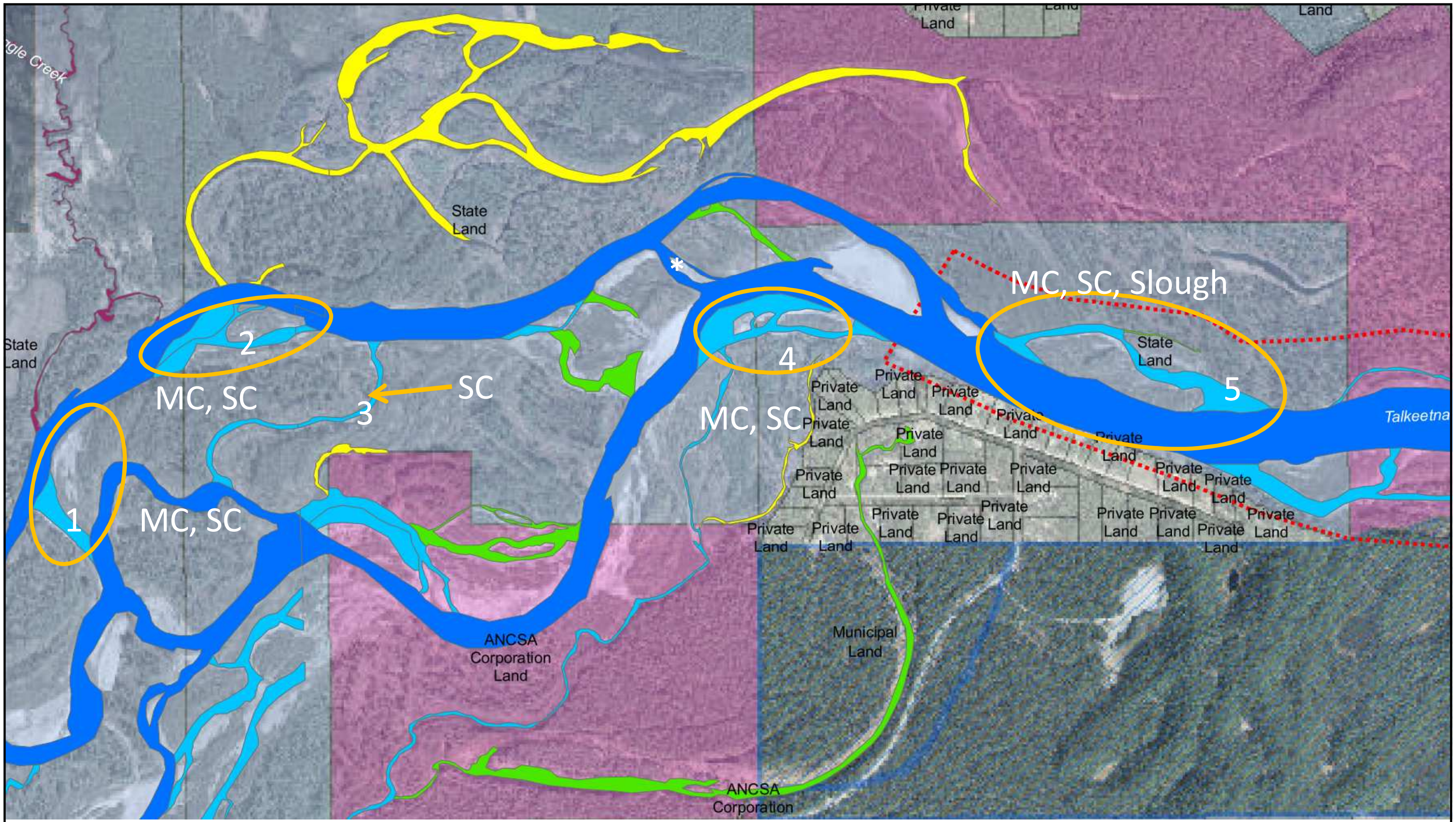


1: Main Channel site, and possible Side Channel site, which looks like it dries up later in summer
2: Main Channel site, and possible Side Channel site
3: Possible Side Channel site
4: Main Channel site, and possible Side Channel site, depending on water levels
5: Main Channel site, and Side Channel site, with a possible Slough site further inland, off the side channel.

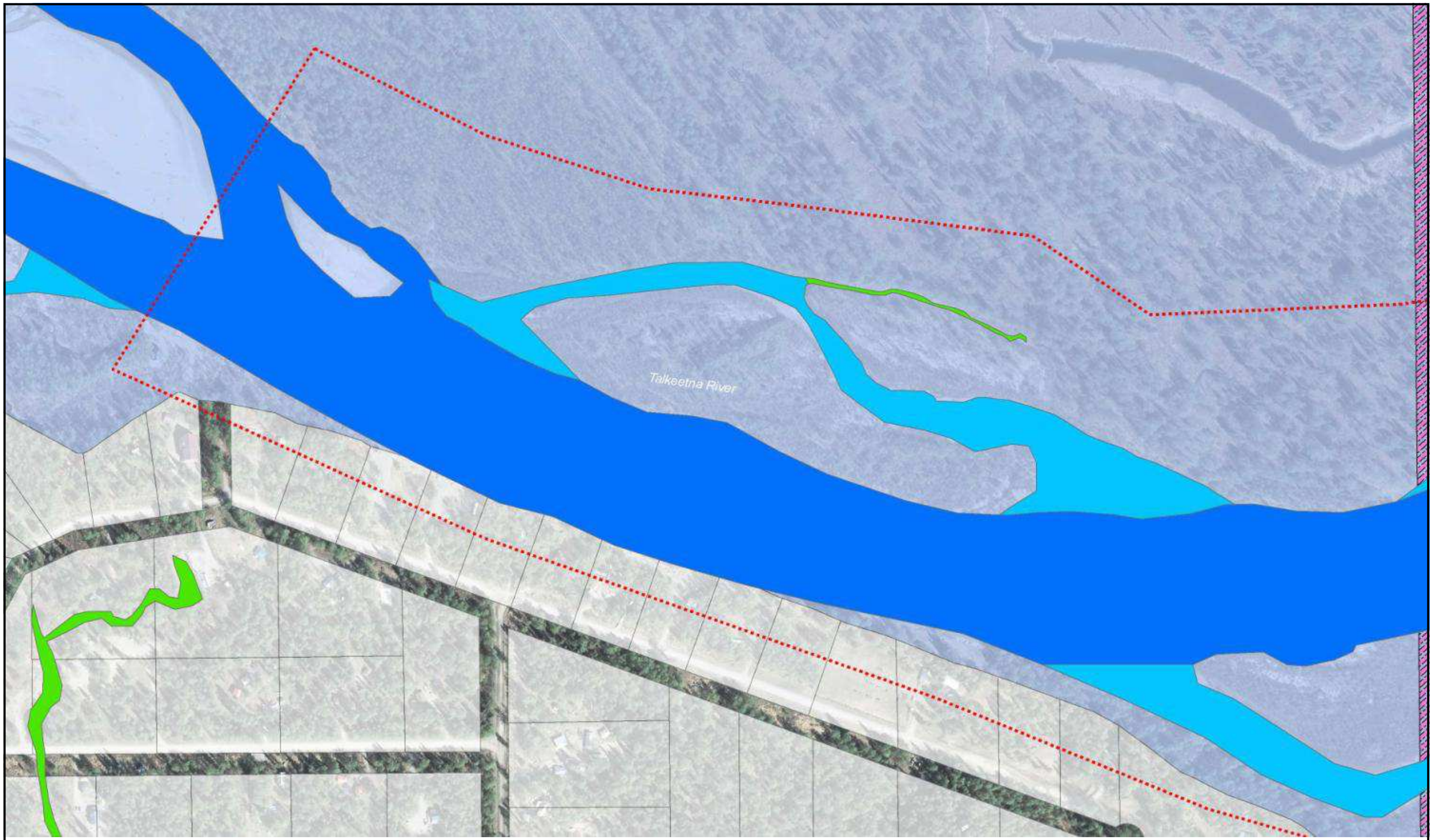
NOTES: Reconn trip stuck to the outer main channel route (the one to the left). Lots of main channel cobble shoreline available. Side channels look to dry up and disconnect for summer, so could be difficult to find one actually running. Area 4 was a wide cobble bar. Area 5 is the preferred, as it has an extensive MS cobble area. Its SC is slower, and a little muddy, and may dry up or become disconnected pools. Slough area is questionable…is it an upland or side slough? Looks like an upland. Measures 400 ft long on the map.



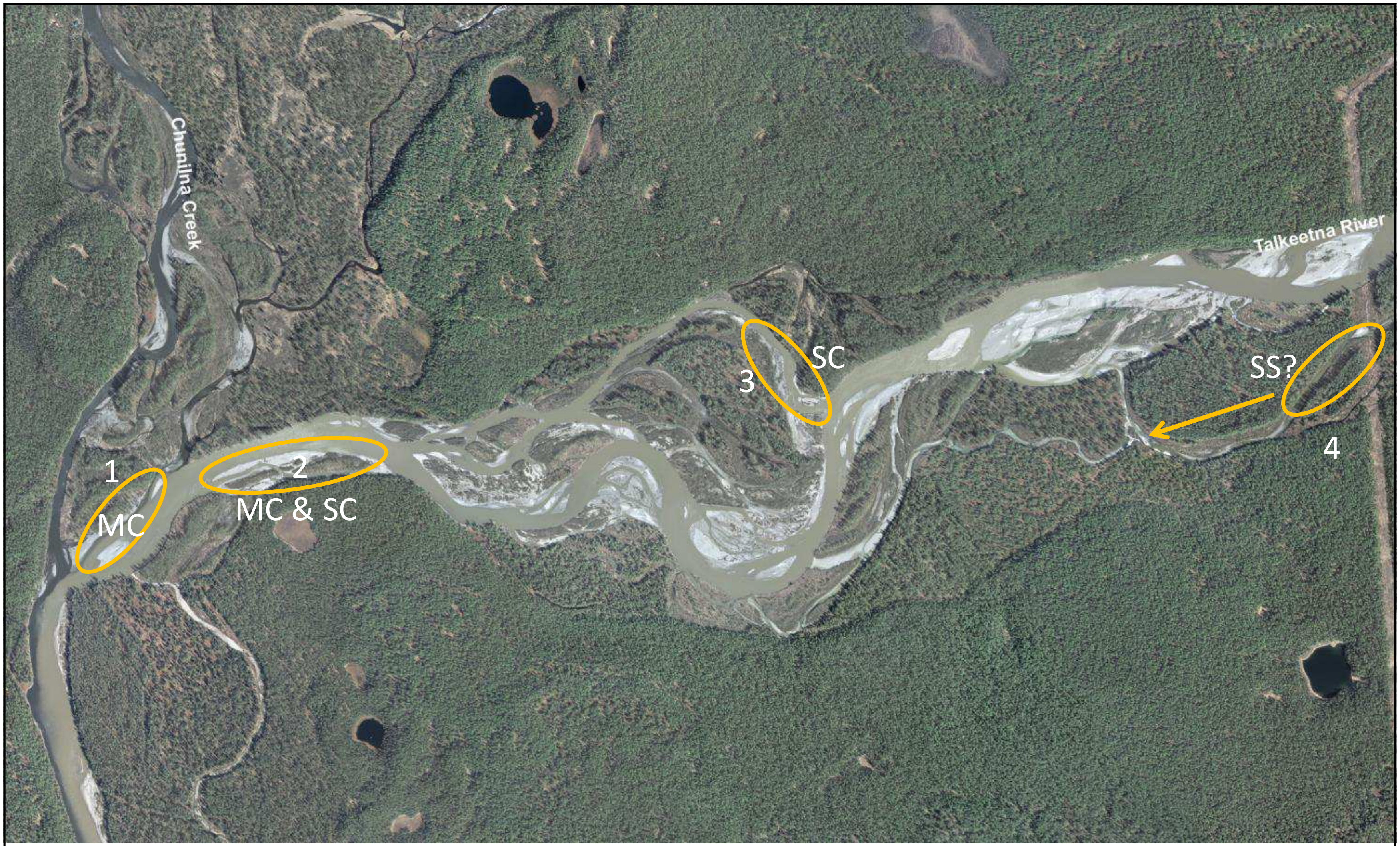
NOTES: Side sloughs near #4 were not apparent in reconn. River has rerouted considerably in 2012-13. The starred (*) MC area is now the main channel route taken by boats.

The side slough running from private land, to municipal, to ANCSA lands is far removed from the main channel, and would not serve as a good comparison to anything we are sampling in the Susitna.

- Area 5, Lower Talkeetna

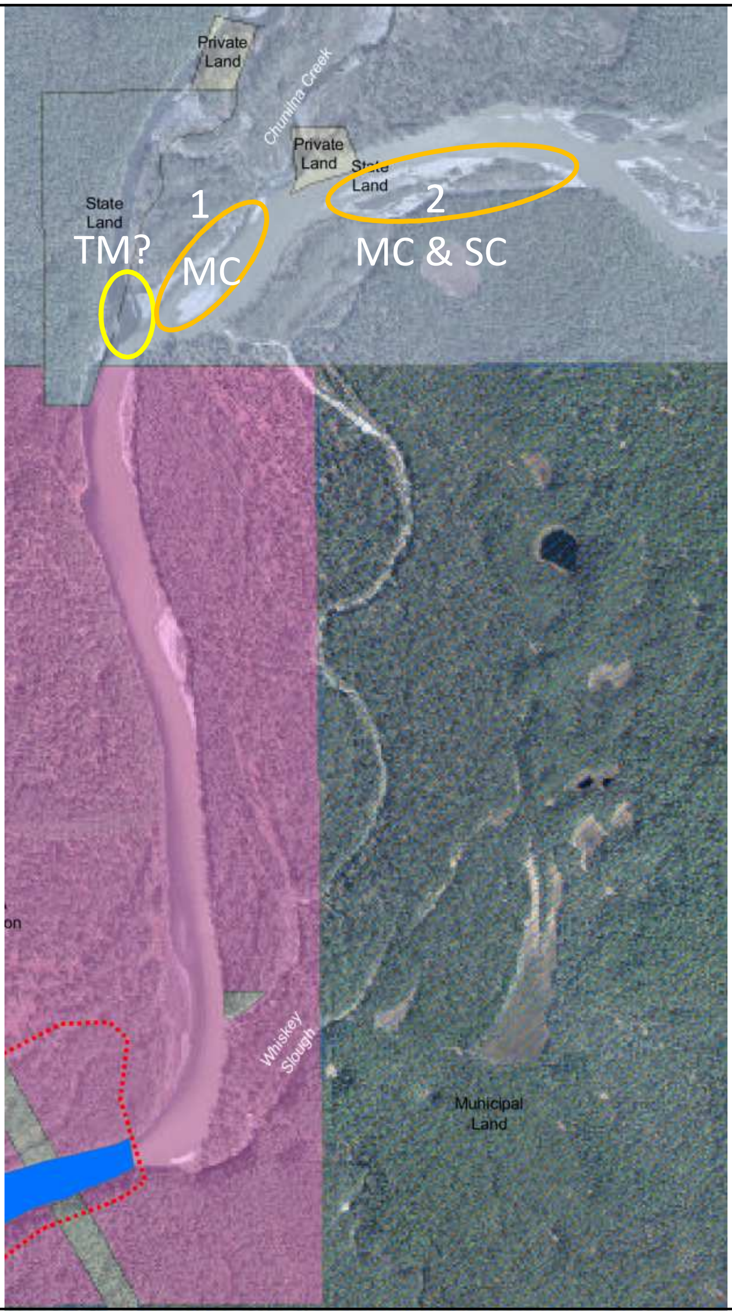


- Area 5, Lower Talkeetna



1: Main Channel site
2: Main Channel site, and possible Side Channel site, depending on water levels
3: Possible Side Channel site, although reconn showed it to be very slow and muddy back there
4: Possible side slough site via power line trail access; extensive network, multiple d/s ends. During reconn the d/s ends of this network appeared to be slow, backwater, muddy areas.

NOTES: Reconn in this reach above Clear Creek stayed on the main channel. Lots of main channel cobble shores to choose from, side channels/sloughs were not explored, but were noted.



NOTES: Slide shows ANSCA lands, where land access is not permitted. This prevents us from sampling the same main channel area sampled by the USGS in 1998 (OFR 99-459, Frenzel and Dorava 1999).

EMAIL RECORD

From: Susan Walker - NOAA Federal [susan.walker@noaa.gov]
Sent: Monday, July 15, 2013 5:00 PM
To: tnightengale@r2usa.com
Cc: Jeffrey Davis; Betsy McGregor; Steele, Marie C (DNR); Berg, Catherine; Buntjer, Michael; MaryLou Keefe; Matt Cutlip
Subject: Re: River Productivity Consult with Agencies on Talkeetna Reference sites

Excerpt from email:

We offer the following preliminary comments on the proposed site selection:
1. Sampling sites should be selected to provide long-term reference measures that can be used to estimate potential long-term, post-project effects.
2. It is unlikely that community composition or macroinvertebrate metrics from any two locations will match exactly. Given the short turn around, I have not had time to review the RSP and PSP; however, the statistical methods that will be used to determine if the Talkeetna sites will provide “feasible” reference sampling locations should be clarified. Given current methods, I believe that there are only 3 replicate samples collected within each macrohabitat. AEA should specify how they will test for differences in mean metrics with a location on the Susitna to evaluate feasibility; will a chi square or other statistical test be used to test for differences in means?
3. Reference sites should replicate characteristics of treatment sites (similar classification); however, geomorphic classification is not provided, and remaining classifications are incorrect or missing.
4. Proposed sites do not provide geomorphic reach classification (slope, confinement, substrate) for these two locations or describe which geomorphic reach on the Susitna would be comparable.
5. The macrohabitats classification shown in the PowerPoint figures is very concerning given the level of discussion we have had on this topic. Clearly there is no main channel habitat in Sites 1, 2, 3, or 4 of Figure 1. These locations area all either split main channel, multiple split main channel, or side channel habitats.
6. If the Geomorphic Reach characteristics are similar to those near MR-8 on the Susitna, the main channel sampling locations noted as Site 5 on Figure 1 opposite Beaver Road should be suitable reference for the similar main channel site opposite Whiskers Creek.
7. As stated in the RSP and repeated in Tim’s email, sampling would occur in one mainstem and two off-channel habitat types. We believe that all 5 macrohabitats should be replicated, as the Services recommended for the Susinta River which FERC supported and we recommend expanding the site selection accordingly.
8. Per AEAs habitat classification, mainstem habitats include main channel, side channel, split main channel and multiple split main channel. Off-channel habitats include side sloughs and upland sloughs. The PowerPoint only shows one Slough in the Wiggle Creek area (Slide 1) near Site 5, AEA has not identified this as a side slough or upland slough, and does not recommend sampling at that location, we concur.
9. AEA still needs to propose some side slough and upland slough habitats within the same geomorphic reach for consideration. There is a large upland slough complex north of the side channel upstream of the mouth of Wiggle Creek above Site 2 of Figure 1. We believe there is side slough habitat on the left bank; but a little more effort may be necessary to identify good sampling locations.
10. If AEA proposes the Clear Creek/Fish Creek area there appears to be upland slough above the side channel Site 3 in Figure 5 and side slough habitat on the opposite bank. (Jeff is available to assess these areas if desired - the Services would support this provided it is within current budget scope and funding.) However, geomorphically, this reach appears more like MR-6. We don't believe AEA has any river productivity study sites in MR-6 therefore this site would not be useful unless AEA

were to add a comparable River Productivity site in MR-6, in which case the study's value for long-term references to estimate potential post-project effects would be increased.

Date: August 5, 2013
From: Timothy L. Nightengale, Kevin Petrone, and Jude Simon – R2 Resource Consultants, Inc.
To: Betsy McGregor – AEA
RE: Account of events - Talkeetna Reference Site selection Field Trip, July 16th 2013

On July 15th, R2 selected and sampled a side channel (SC) site and located a potential upland slough (US) site located across from the Talkeetna River Guide put-in (Lower Talkeetna River).

On July 16th 2013, staff from R2 Resource Consultants (Kevin Petrone, Jude Simon, and Benjamin Meyer) met with Jeff Davis (ARRI; Services' contractor) to consult on the selection of three Talkeetna reference sites for the River Productivity study. The sites were to include one main or side channel site and two off-channel sites and, according to Mr. Davis, were to be located within a geomorphic reach that is similar to either the Whiskers or Indian Focus Areas.

The group visited the following sites in the lower Talkeetna River (below the USGS gaging station):

- The side channel site sampled by R2 on July 15th
- An upland slough and side slough sites opposite the Talkeetna River Guides put-in.
- An additional side slough site across the river from the upland slough and side slough sites.

From these sites, it was determined:
- The side channel site was acceptable as a main channel habitat site;
- The side slough sites was not connected with the river and the upland slough site had too little flow;
- Neither the upland slough nor side slough were comparable to the Whiskers and Indian River slough sites; and,
- The additional side slough site was not adequate because it lacked groundwater flow.

The group continued the reconnaissance by examining potential sites upstream of the Clear Creek confluence. The two-hour effort located two off-channel site locations:
- A suitable side slough site with groundwater upwelling;
- An upland slough with beaver activity

Regarding the upland slough, the group discussed the following:
- The site might be suitable as an upland slough site, reasoning that the upland slough site established at FA-104 (Whiskers Slough) was also beaver influenced; however, the upland slough site at FA-104 (Whiskers Slough) is established upstream of a beaver dam and beyond the influence of the beaver as indicated by a cobble substrate.
- After further discussion, the upland slough site was retained as a Talkeetna site, because another suitable upland slough site in close proximity to the selected side slough site could not be found.
- In order to establish a sampling station, with all sampling sites located in the same local area, the side channel site sampled on July 15th was discarded, and a new side channel side was chosen as the main channel habitat site, near the upland slough site location.

After sampling the US site, R2 confirmed that the upland slough site did lack hard substrates that are being used for collecting algae samples at other sites. Although sampling from woody debris and soft substrates is possible, the volume of silt in these samples means that the samples are processed and analyzed differently at the lab. R2 will need to review the data after processing to evaluate if these data collected via different methods will be comparable with the samples collected at the sites established on Middle Segment Susitna River. R2 sampled the remaining SS site and a new mainstem site in the same area as the US site over the next two days.