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The Effectiveness of Forested and Hedgerow Riparian Buffers for Buffering Water Temperature and Improving Fish Habitat in Agricultural Waterways in Western Washington

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THE EFFECTIVENESS OF FORESTED AND HEDGEROW RIPARIAN BUFFERS FOR BUFFERING WATER TEMPERATURE AND IMPROVING FISH HABITAT IN AGRICULTURAL WATERWAYS IN WESTERN WASHINGTON

By

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Abstract

Riparian restoration is a component of nearly every salmon recovery strategy. In the lowlands of the Nooksack River flood plain in Western Washington State, planted riparian buffers in agricultural landscapes must perform multiple functions to improve water quality and fish habitat while still allowing access to agricultural land use. Relatively narrow, 15 feet (4.6 meter) wide buffers, are a more palatable option for landowners than 35 feet (10.7 meters) which is required to be considered for cost incentive programs such as the Conservation Reserve Enhancement Program (CREP). We wanted to discover whether these two relatively narrow buffer widths would result in detectable differences in the effectiveness of water temperature maintenance (reduction from upstream to downstream warming via % effective shade) and differences in fish assemblage abundance. Results from this research conducted in 2014-2015 indicate that in 100-m long reaches, narrow, 15-ft-wide buffers provide similar amounts of shade as wider, 35-ft- wide buffers, but differences in upstream and downstream water temperature in terms of heat units were inconclusive. Heat units were expressed as the daily cumulative degrees above 17.5°C relative to the number of temperature readings each day (Biologically Sensitive Heat Units). This excluded several sites from analysis that never reached temperatures above 17.5°C and temperature maintenance at the remaining sites were highly variable. Differences in width was not a significant factor detecting differences in relative abundances of fish communities, but the 15’ and 35’ sites had greater species diversity and greater abundances of native coldwater species, such as coho salmon and cutthroat trout, than the sites without buffers.

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Introduction

Riparian buffers in agricultural landscapes: a literature review

a. Overview

In this study I wanted to know what effect narrow buffers in agricultural landscapes have on water temperature and fish communities at the local reach scale. The purpose of this study was to determine whether buffer width 1) was a controlling factor for shade levels over the agricultural waterway, 2) allowed for water temperature maintenance from an upstream to a downstream location over 100 meters, and 3) was related to detectable differences in relative abundance of fish assemblages. I begin by giving context to where and how riparian buffers are established and their purpose in agricultural areas. Later I report recommendations for buffer widths that achieve desired outcomes in terms of shade, water temperature, and fish habitat. And, lastly, I give background on factors which may affect shade, water temperature, and how fish populations respond to riparian buffer establishment

It is generally accepted that establishing vegetation around aquatic habitats to create a “buffer” between the waterway and human land use is a beneficial practice, and so it is employed worldwide (Roni 2008). Riparian buffers provide ecological functions such as shade, slowing overland runoff, and intercepting pesticides and other aerial pollutants and providing habitat for stream biota (Young et al. 1980, Zwieniecki 1999, Duval and Hill 2006, Mankin et al. 2007; Zhang et al. 2010).

Riparian buffers are established through regulation and implemented through voluntary programs, namely the Conservation Reserve Enhancement Program (CREP). Planted buffers exist in prescribed widths according to standard practices that are set by the Natural Resource

Conservation Service (NRCS) and are implemented by Conservation Districts across Washington State. Typically, buffers are 15 feet, 35 feet, or a width between 35 and 180 feet on each side of a waterway, which is the maximum width that can receive funding through CREP. After establishment, these buffers are monitored a few years for plant survival and the amount of shade they produce over the waterway. Much less is known about how closely the ecological functions they provide mimic natural conditions.

The impetus for evaluating buffer effectiveness begins with environmental policy at the state and federal levels. As buffers are evaluated, the consensus among studies constitutes the "best available science" and is sought for determining buffer program policy in Washington state (RCW 36.7OA.172). Environmental regulators then apply knowledge from research to create plans for environmental protection according to a desired outcome. Policy directs buffers to be set at fixed widths because a common set of dimensions is easier to prescribe than analyzing site specific conditions in each location (Castelle et al. 1994).

While there is extensive research on buffer effectiveness on land used for timber harvest, less has been completed on the densely planted narrow conservation program buffers in agricultural lands and their effectiveness, including in Whatcom County. In addition, monitoring and systematic evaluation of the width of a planted buffer as it relates to effectiveness for habitat over more than two to four years are severely limited (Paulsen and Fisher 2005, Roni et al. 2014). Most commonly, the presence of buffer vegetation-rather than specific width- has been related individually to shade, water temperature, or fish assemblages. For example, an early report of Washington CREP projects records data from a 5-year period on statistics of plant density, diversity, survival, growth, percent of canopy shade, total area of the planting, and the type of ESA-listed species potentially benefitting from the project (Smith 2006). One follow-up report from the Conservation Reserve Program for the State of Washington describes planting survival and a 10-site quantitative analysis of the mean percent effective shade provided by buffers 4 to 10 years old but does not consider variations in width (Smith 2012). In Virginia, CREP monitoring programs used the Stream Visual Assessment Protocol (SVAP) to quantify changes in riparian vegetation and correlate them with fish assemblage health (Teels et al. 2006). The SVAP is a more in-depth tool than records kept in Washington State, since its score is the mean taken from scores of individually assessed elements such as channel condition, hydrologic alteration, riparian condition and width, bank stability, water appearance, nutrient enrichment, barriers to fish movement, in-stream fish cover, pool quality, canopy cover, manure presence, riffle embeddedness and invertebrate habitat. Teels et al. (2006) found that the most disturbed sites, or areas with initial high scores on the Human Disturbance Index, improved their SVAP score the greatest one year after buffer establishment, but only 42.4% maintained scores higher than the baseline. However, none of the changes were related to any specific buffer width. Some grass roots organizations such as the Tenmile Creek Clean Water Project Committee collect water temperature data, but the buffer widths at sample sites are variable (Belisle et al. 2008). The most in-depth study in the Puget Sound Region (WSU/ UW 2008) estimated the abundance of salmon and trout in agricultural waterways in King County but did not compare among buffer widths.

Non-point source pollutants originating from agricultural land are considered one of the primary causes of water quality degradation (EPA 2004). Washington state requires each county to develop land use plans to protect its local natural resources which include water quality (WAC$_a$ 365-196-485). For example, in Whatcom County farm plans are required for small farms with moderate to high contribution of nutrient and sediment runoff to a "Critical Area". Critical areas are ecologically sensitive areas that include riparian zones (WCC 16.16.290). Farm plans describe the methods the landowner will employ to prevent runoff from entering water bodies and often include establishing riparian buffers. Similarly, large livestock operations are required to obtain a permit from the Washington State Department of Agriculture and comply with required buffer widths for preventing waterway contamination. Besides ensuring compliance with farm plans and water quality permitting, benefits to farmers from the riparian buffers include reduction in topsoil loss and improved drainage by reduction of dredging (Castelle et al. 1992, Dosskey 2001, Yuan et al. 2009, Arora et al. 2010, F. Corey, Whatcom Conservation District, Personal Communication, August 2015).

Restoring water quality for the abundance and health of fisheries resources is a primary objective in the Pacific Northwest, after maintaining water quality for public health (EPA 2003). Research identifying gaps in fisheries management strategies and techniques for assessing habitat has revealed that a lack of inland freshwater habitat and impaired water quality are among the limiting factors for Pacific salmon and trout species (Smith 2002). This is especially important in Whatcom County, Washington, because much of the waterways linking salmon to their spawning and rearing grounds pass through agricultural areas. So, planting riparian buffers in agricultural lands is also a direct way to improve fisheries habitat.

There has been much effort to synthesize the literature to determine a prescription for an adequate "effective" buffer width for mitigating effects of forest harvest and agricultural practices. A review by Castelle et al. (1992) found a range of 3 to 200 m to be effective. Sweeney and Newbold (2014) recommended 30 m to protect streams in multiple aspects: fish, macroinvertebrates, nitrogen, erosion, temperature and large woody debris. Haberstock et al. (2000) recommend a 36-m forested buffer for the protection of Atlantic salmon habitat. Others recommend widths for specific buffer functions rather than an umbrella of ecological functions. Wenger (1999) provides buffer width recommendations according to the function of the buffers, i.e. sub-surface nitrogen removal (15-30 m for >90% removal), sediment removal efficiencies (9-30 m for >90% removal), temperature (at least 10 m), and woody debris (15-130 m). A meta-analysis by Mayer et al. (2005) showed that 90% nitrogen removal was possible with a 149-m wide buffer. In general, recommended widths vary greatly depending upon the type of pollutant or land use to be mitigated.

Buffers in agricultural landscapes must perform multiple functions to effectively mitigate the adverse effects to the natural environment caused by farming while providing wildlife and fish habitat. Several studies document buffer efficiency in reducing bank erosion (Dosskey 2001; GEI Consultants 2004), creating a barrier to pesticides (Vought et al. 1995, Borin et al. 2004, Arora et al. 2010), filtering nutrients and sediments from runoff (Borin and Bigon 2002, Yuan et al. 2009, Zhang et al. 2010) and acting as protective corridors linking otherwise fragmented patches of habitat (Fremier et al. 2015).

In Washington, agricultural drainage waterways in lowland Puget Sound are being increasingly managed for fish and are considered potential areas for re-vegetation of former riparian habitat. Agricultural waterways are maintained to drain water from the flood plain and to provide access to use of water for irrigation. Buffer establishment sometimes conflicts with the needs of agricultural practices, but concerns can be addressed. Conflicts of farming with buffers are outlined by Jia et al. (2006) and include conversion of profitable crop land for buffer establishment, reduction of field shape and size, and reduced accessibility of the stream for irrigation equipment setup (pump stations) and machine maneuverability (irrigation guns with hoses). Paying for easements within CREP to offset financial hardship and allowing plantings with gaps to facilitate irrigation equipment access are methods for preserving a harmonious balance between industry and conservation.

In Whatcom County, a major impairment to drainage and native fish habitat is thick, homogeneous stands of reed canary grass (*Phalaris arundinacea*) that dominate un-buffered agricultural drainage waterways (Figure 1). Accumulation of the dying grass in a waterway lowers the concentration of dissolved oxygen (DO) to levels that adversely affect fish. When the grass decomposes, the biological oxygen demand increases and DO levels can fall below 5 ppm (Milburn 2007). Over time, the deposition of decaying reed canary grass also reduces waterway flow. This impedes use of fish habitat and functionality of the waterway for farmers.

A typical agricultural practice is periodically dredging clogged waterways to improve flow. It can increase dissolved oxygen levels over a short period (Milburn 2007), but dredging is expensive and may require multiple permits and oversight from WADE and Washington Department of Fish and Wildlife (WDFW). Riparian buffers with trees or densely planted shrubs provide enough shade to prevent reed canary grass establishment (Tu 2004), thereby reducing the amount of plant material clogging drainage waterways.

Lessening the frequency of agricultural waterway dredging would benefit fish populations through less disruption of bank and instream habitat (Chapman and Knudsen 1980).

b. Riparian Buffer Width Effectiveness on Shade and Temperature

Shade and water temperature are linked since one of the most influential predictors of water temperature warming in lowland streams as shown to be shade over the waterway (Mayer 2012). In terms of shade provision, effectiveness of riparian buffers varies not only by width, but depends on a multitude of local site characteristics: latitude, stream aspect, leaf area index (density), and vegetation height from the water surface (Sridhar et al. 2004, Dewalle 2010). Annual maximum stream temperature depends on the proportion of shade over the stream, proportion of the watershed with woody vegetation cover, elevation, hydrologic inputs, geomorphology, tile drain presence, distance to the ocean, air temperature, regional topography, and solar radiation (Zwieniecki and Newton 1999, Johnson 2004, Mayer et al. 2005, Tague et al. 2007, Rex et al. 2012, Chang and Psaris 2013). At the local scale, the proportion of shade over the stream is the greatest influence on longitudinal changes in temperature within a stream (Chang and Psaris 2013).

The goal of most research completed at the local scale aims to define the minimum buffer width that provides the greatest effect for minimizing changes in longitudinal water temperature (Table 1). Many of these reviews and studies were conducted at high elevation, in areas dominated by forested land use, with taller maximum tree heights than planted agricultural buffers. Some reviews give recommendations of buffer width that consider effectiveness for multiple buffer functions. From studies that only considered shade and water temperature maintenance, effective buffer widths fall between 10 m and 30 m (Wenger

1999, Zwieniecki and Newton 1999, Broadmeadow and Nisbet 2004; Hawes and Smith 2005, Wilkerson et al. 2006). Several studies maintain that the density of the buffer is directly related to effectiveness of shading when width varies (Castelle et al. 1994, Haberstock et al. 2000, Broadmeadow and Nisbet 2004, Sridhar et al. 2004).

From forested and agriculturally dominated landscapes, there is evidence to suggest that narrow (4-15 m) buffers are effective at providing a high proportion of canopy cover (Dewalle 2010). Dewalle (2010) described a theoretical model that revealed that a 12-m-wide buffer 30-m-tall with a Leaf Area Index (a quantity used to describe vegetation canopy as leaf area per unit ground surface area) of roughly 6 would provide 80% shade over a stream 3 m wide. One review concluded that narrow (defined there as less than 10 m) buffers can reduce overland flow and provide shade but also found that few studies in Washington measured the effectiveness of narrow widths (GEI 2004). Ryan et al. (2013) found that one to two rows of trees provided enough shade over small streams (<4 m wide) with a granite bedrock substrate to prevent longitudinal stream warming by blocking solar input. This made it possible for the stream temperature to decrease by 1˚C over 300 m because of other environmental cooling interactions. When considered primarily for shade contribution, narrow buffers seem to provide an effective remedy to bare agricultural ditches.

Variation of local conditions can account for the wide variation in buffer width recommendations. Additional information is lacking in most studies, but it has been suggested that these key variables, density of vegetation, initial upstream temperature conditions (Barton et al. 1985), and groundwater inputs (Harper-Smith 2008), help explain buffer effectiveness. Without these details the results of studies of width and shade effectiveness on stream temperature could be confounded. Streams have a natural warming trend from headwaters to river mouths, and background warming rate is not always accounted for in studies. Water temperature change could be as subtle as less than 0.18˚C per 152.4 m (Cristea & Janisch 2007).

Effectiveness of buffer width on stream temperature is much less clear-cut than measuring the percent shade under the canopy of a given stream reach. Often an intermediate effectiveness in water temperature maintenance is achieved with narrow buffers. A review by Sweeney and Newbold (2014) found the average increase in water temperature of un-buffered reaches was 5˚C over 100 m reaches. Only two of the 17 cited studies examined buffered site widths between 0 and 10 m, and both resulted in an increase of water temperature greater than 1.2˚C (Hewlett and Fortson 1982, Davies and Nelson 1994). Both studies were located in timbered forest land. The narrowest forest buffer width that maintained no change in temperature was 10 m (Sweeney and Newbold 2014). Other research found that narrow buffers maintained low daily maximum temperatures, which is similar to the effectiveness of larger buffers (Zwieniecki and Newton 1999). Mature forest buffer 8.6 to 30.5 m wide prevented an increase in warming outside of the natural warming trend measured in control reaches (Zwieniecki and Newton 1999).

This literature review focused on buffer establishment only for the purposes of providing shade and minimizing increases in water temperature. Buffer width recommendations are wider for functions that are required when the goal is also to include structural habitat for salmon and trout. Functions such as pool creation from large woody debris recruitment and longevity of the buffer are concerns in upland forested areas. Wide buffers (i.e., >23 m) are recommended in forest harvest areas to provide long-term durability of buffers. For example, in higher elevation areas of forest harvest associated with tall tree heights and steep slopes, wider buffers account for buffer blowdown likely to occur at the buffer's edge (Pollock and Kennard 1998). Haberstock et al. (2000) found that maintaining ample shade, stream flow, coarse woody debris and sediment filtration (important to Atlantic salmon habitat) would be possible with a buffer of 36 m. This was estimated using a model to select a variable width, zoned approach appropriate for site specific conditions.

c. Riparian Buffer Effectiveness and Fish Abundance in Agricultural Waterways

Fish as Measures of Habitat Effectiveness

Fish communities are used to indicate ecosystem health and habitat disturbance (Wichert and Rapport 1998). At the physiogeographic scale in the Pacific Northwest the fish Index of Biotic Integrity is used to assess aquatic ecosystem condition (Mebane et al. 2003). It scores condition based on 10 metrics, some of which are number of native coldwater species, proportion of sensitive native individuals, number of coldwater individuals, percent degraded water quality tolerant individuals, and number of aged classes of salmon and trout (Mebane et al. 2003). Assessments of this nature are ideal for comparing environmental conditions over long periods of time, as populations respond slowly over wide geographical areas to habitat change. In general, increasing the complexity of in-stream structural and vegetative habitat alters fish diversity and composition and increases juvenile survival (Paulsen and Fisher 2005, Smokorowski and Pratt 2007). When regional land use changes including riparian buffer establishment in an agriculturally dominated watershed were compared over 43 years by Wichert and Rapport (1998), they found improvements to fish communities.

At the local scale, differences in fish assemblage relative abundances, fish abundance, and biomass are good indicators of habitat effectiveness because fish select habitat quickly according to changes in water temperature during summer (Hillyard and Keeley 2012, Armstrong and Schindler 2013). In Ontario, Canada, Stammler et al. (2007) found no difference in fish relative abundance between agricultural buffers and reference sites, but control and agricultural reaches were within an equally degraded agricultural watershed. Walser et al. (1999) found that habitat complexity of agricultural land was not correlated with species abundances for headwater streams. Both of these studies occurred in warm-water systems or at the upper thermal margins of trout habitat. There is no peer-reviewed literature comparing relative abundance of salmon and trout in different buffer widths of agricultural waterways in western Washington. Existing data about fish habitat use are in the form of surveys collected by state agencies during visual surveys or while supervising watercourse management activities, e.g., dredging and culvert construction (Berge 2002, WCCD 2008). Information gained from this study was used to evaluate whether narrow buffers were adequate for providing fish habitat in a lowland cold-water system.

Influence of Environmental Factors on Salmon and Trout Presence and Abundance

To test for differences between various buffer widths, relative abundances of fish communities were used in this study as a response variable. Presumably, as buffer width increases, there would be a proportionate addition of larger wood to add complexity to the site and therefore they would provide better habitat for salmon and trout. I assumed more complex habitat would aggregate fish into greater relative abundances at sites with the widest buffers. Other factors besides buffer width determine the presence and abundance of salmon and trout. Physical access to habitat withstanding, water temperature is the primary limiting factor affecting the presence and abundance of juvenile salmon and trout in agricultural drainages. Channelization cuts off access to the flood plain and with it potential cooling influence of groundwater upwelling and alter water flow rates. Lack of woody vegetation exacerbates temperature rise. Both channelization and sparse vegetation affect the accessibility of spawning sites upstream, rearing habitat, and invasive fish species emigration (Chapman and Knudsen 1980, Zika and Peter 2002, Colvin et al. 2009, Pollock et al. 2009, Andrew and Wulder 2010). The degree to which salmon and trout are affected by these factors vary by species and their particular needs at different stages in their life cycles. The following examination of habitat requirements of salmon and trout in agricultural waterways focuses on their needs at the juvenile stage during summer months.

Access and Availability of Habitat

Availability of habitat is a controlling factor for fish presence and abundance of a species; as demonstrated in the comparison of coho salmon abundance in forested streams before and after clearcutting and large wood removal (Bisson and Sedell 1984 in Maser 1988). Habitat disturbances can have a domino effect on multiple stages of salmon and trout life cycles. Adverse water quality conditions and physical channel obstructions are the two principal barriers to accessing upstream spawning areas that can limit fish habitat use in lowland streams (Price et al. 2010, Fenkes et al. 2016). In this study, all sites had similar water quality conditions and were not obstructed by physical barriers to movement of juveniles. But, we do not know if there were spawning sites upstream of all sites. The quality of the spawning site habitat determines the success of the offspring especially for those species that rear in slow moving waters for a year or more, i.e. coho salmon, Chinook salmon, and rainbow trout.

Water Temperature

Maximum stream temperature is one of the most important factors in determining trout presence because of its adverse effects on survival (Barton et al. 1985). Water temperatures of more than 20˚C (Smith 2002) and dissolved oxygen levels more than 8 mg/L (Bjornn and Reiser 1991) limit their populations by causing sub-lethal stress and mortality. Riparian vegetated areas reduce water warming from solar radiation and diel maximum fluctuation in water temperature (Malcolm et al. 2004). Temperatures for rearing trout and salmon should not exceed a 7-day maximum mean temperature of 17-19 ˚C based on an extensive review of the literature by Washington Department of Ecology (WADE 2002, WADE 2012). In juveniles, temperatures above 18.5˚C reduce metabolism by slowing conversion of energy to biomass and thus growth is reduced (Sullivan et al. 2000). Higher fluctuation in day to day temperatures also reduces salmon growth (Willey 2004). In this study, the accumulation of heat units above 17.5˚ C is assumed to be detrimental to juvenile salmon and trout rearing conditions based on 17.5˚ C being the criterion limit used by Washington State for the 7-day average of the daily maximum water temperatures suitable for salmon rearing and migration ($\mathrm{WAC_b}$ 173-201A-200).

Shade

Vegetation and the shade it provides maintain conditions that create thermal refuge and create areas of cover for small fish to avoid predation and for adults waiting to spawn

(Bjornn and Reiser 1991). The amount of cover in the landscape can be a predictive variable for status in salmon populations in species such as coho salmon and Chinook salmon (Maret et al. 1997, Andrew and Wulder 2010). At this scale, proportion of cover also corresponds to reduction of sediments and increased pool creation from woody debris additions.

At a local scale, fish response to shade varies depending on whether particular habitat is light-limited- meaning that shade restricts phytoplankton growth and prevents water from warming to optimal temperatures for fish growth (McCormick and Harrison 2011). If water temperature is consistently below temperatures that allow optimal growth rates when food is plentiful, fish will seek sections of waterways that are unshaded and warmed by solar radiation. Conversely, when water temperatures are near or rise above the threshold that allows optimal growth rates, salmon and trout will seek out cooler shaded stretches or deep pools to find refuge. Fish response to shade also varies by season (Koski et al. 1984, Platts and Nelson 1989). Riparian vegetation providing overhead bank cover and shade explained 31% of the variance in trout biomass per area in Wyoming (Wesche et al. 1987). In the Pacific Northwest, juvenile salmon growth can be limited by their food source of macroinvertebrates that feed on periphyton. Periphyton decreases as vegetation begins to shade the stream (Koski et al. 1984). Salmon and trout populations may respond positively in summer to un-vegetated waterways in cold climates because stream productivity and water temperatures are higher, and thus closer to preferred growth temperatures, compared to shaded stream sections (Koski et al. 1984, O’Grady 1993). During winter, Chapman and Knudsen (1980) found that the biomass density of streams was reduced but observed no difference in numbers of age-0 trout between open sites and those with riparian vegetation.

Furthermore, in the Willamette Valley, OR, an area known for its turf grass production, buffers without woody vegetation can provide habitat for fish in winter if there is grass present (Colvin et al. 2009).

Sedimentation

Sedimentation disrupts the behavior of juvenile coho salmon at 60-70 NTU (nephelometric turbidity units) (Bisson and Bilby 1982, Berg and Northcote 1985, Bjornn and Reiser 1991). When avoiding faster, more turbid water conditions of large streams, juveniles use agricultural waterway side-channels with clear water conditions. Maintaining these conditions is important to juvenile survival.

Instream Cover and Channelization

Instream cover (pools, woody debris, and substrate type) is important to the presence and abundance of juvenile salmon and trout because habitat abundance is a limiting factor for the carrying capacity of the stream. In particular, Coho salmon biomass is related to pool habitat with stable large wood (Maser et al. 1988). Agricultural waterways are generally devoid of instream cover because logs, rocks, and roots can slow water drainage from fields.

Channelization severely simplifies habitat and limits cover by removing thermal refuges. Chapman and Knudsen (1980) found that in western Washington, channelization significantly reduces the quality and quantity of habitat for adult coho salmon and cutthroat trout over time. They observed a decrease in adult trout biomass per square meter, though age-0 trout biomass increased, but the overall biomass of all ages of trout and coho declined (Chapman and Knudsen 1980).

In areas with permeable soil, subsurface drains increase the intensity of runoff during rainfall events and reduce ground water input during dry conditions (Blann et al. 2002). Because of the hydrologic regime changes in agricultural lands using surface and subsurface drains, fish communities tend to gradate toward tolerant, generalist species (Blann et al. 2002). Tolerant species are those that can survive in conditions with increased sediment and chemical pollutants, “flashy hydrographs”, and altered patterns in water temperature (Blann et al. 2002). Many times tolerant species are non-native species adapted to warm-water conditions. Channelization and drainage indirectly cause competition between native cold water and introduced warm-water species and is a concern (Barton et al. 1985).

Thermal sensitivity describes how quickly a stream warms or cools, and affects how much temperature fluctuates. A stream’s thermal sensitivity is controlled by ground-water input, channel dimensions, watershed size, and distance to the ocean (Chang and Psaris 2013). Agricultural waterways are typically homogeneously shaped; narrow, shallow, and channelized. This type of channel morphology is an important controller of stream thermal sensitivity in agricultural areas, i.e. shallow streams warm faster than deep ones (Zwieniecki and Newton 1999). Long stretches of non-shaded, channelized waterways absorb heat quickly, causing sub-lethal temperatures, thereby creating a barrier to adult spawning migration or risk of reduced realized fecundity (Fenkes et al. 2016).

Water Velocity

In early spring slow water conditions are important for salmon and trout species, especially those overwintering in first and second order waterways. Agricultural waterways with buffers are inhabited in the summer by juveniles escaping faster flows of larger tributaries. For example, presence and abundance of juvenile anadromous cutthroat trout is predicted by channel width and is inversely associated with gradient (Rosenfeld et al. 2000).

d. Buffer Effectiveness: What is it and how is it measured?

Riparian buffer effectiveness is measured by examining the relationship between buffer width and how closely the associated stream resembles a natural condition. In research studies, buffer effectiveness is often communicated as percent reduction in nutrients as they move through the soil toward the channel, rate of temperature increase over a length of stream, and indexes of biodiversity and health of life in the aquatic system itself (Lee et al. 2001). For example, Borin and Bigon (2002) found that in a 5 m buffer strip with one row of trees NO_3 concentrations exiting the buffer were 90% of the concentrations measured in the field and did not exceed a total concentration of 2 ppm. In a study by Zwieniecki and Newton (1999), change in stream temperatures were compared 1) from the upstream to the downstream boundary of buffers left after forest harvest, and 2) from a completely forested "recovery zone" that was 150-300 m downstream from the harvested zone. Some warming occurred in very narrow buffered sections but no significant difference was found of a persistent temperature warming trend 300 m downstream.

Buffer effectiveness is determined by assessing numerous factors based on how they relate to a desired outcome. An expected outcome of an effective buffer for habitat restoration is creating fish habitat either locally (shade and wood addition) or downstream (maintaining water temperature). Narrow buffers of 35'(10.7 m) or less are mainly used to shade streams in hopes of creating a microclimate that regulates stream temperature. For this reason, water temperature maintenance and fish assemblages were selected in this study as the metric to be measured for narrow buffer effectiveness.

e. Objectives

This study aimed to measure the effectiveness of narrower buffers for shade, water temperature, and fish habitat during a period of peak summer temperatures in 2014 and 2015. Grants available from Washington State Department of Ecology for establishing riparian buffers for habitat are available to landowners only when a minimum of 35’ (10.7 m) of buffer is established. Establishing buffers takes land out of production and can alter expensive nutrient management plans on farms with animal production. If policy could allow a broader range of functional buffer widths that would be subsidized, then more farmers may be inclined to pursue riparian restoration. However, effectiveness of narrower buffer widths (less than 35 feet wide) have been less studied than wider buffers so it is important to clearly define the ecological functions that narrow buffers provide.

Private landowners have a stake in knowing whether establishing riparian buffers is making a difference in water quality and salmon recovery. I believe the agricultural community of Whatcom County would be particularly interested in ensuring congruency in policy strategy and buffer effectiveness. Although CREP buffers offer funding support to landowners only for buffers of 35 (10.7 m) feet or more. It is important to note there could be benefits to fish from establishment of narrower buffers where there were none before. Therefore, this project was supported by the Washington State University Whatcom County Extension to quantify the effectiveness of narrow riparian buffer widths and extend the findings to the agricultural community.

The objective was to determine the effectiveness (or lack thereof) of narrow buffers on water temperature maintenance and on fish habitat, in Whatcom County, WA. I hypothesized that the temperature from upstream to downstream would be maintained or would decrease in 15-foot (4.6 m) and 35-foot (10.7 m) buffered reaches as compared to temperature increases at reaches without buffers and that this would occur at sites where the threshold of effective shade was greater than 65% (Cristea and Janisch 2007).

The evidence in western Washington for how salmon and trout respond to narrow buffer widths is particularly sparse. I inferred from studies of stream reaches with forested buffers that between 10 and 30 m of forested buffer could provide at least patchy shade above 60% (Table 1) and that fish will generally respond positively to the amount of cover available that is over and within the stream (Table 1). The hypothesis, that fish assemblages would differ according to buffer width, was tested by examining the data for similarities in fish assemblages within site widths and differ among them.

This study sought to answer these questions and add to the body of best available science on whether buffers less than 35’ (10.7 m) wide on each side of the stream in agricultural areas serve as effective shade, maintain stream temperature, and provide fish habitat.

Methods

a. Study Area and Site Selection

The study area was located in Whatcom County, WA in agricultural areas south of the city of Lynden. For identification purposes, the sites were given three letter abbreviations according to ownership and buffer width (Table 2). All sites were within the Whatcom Basin physiographic region which includes the Lowlands of the Nooksack River flood plain that are mostly less than 15 m above sea level in elevation (Goldin 1992). A total of 14 sites were monitored within the Scott, Fourmile, Tenmile, and Deer Creek drainage basins of the Lower Nooksack sub-basin (Figure 2). Three sites were reaches within the Tenmile Creek where flows are maintained by groundwater inputs in its upland reaches (Goldin 1992). It is possible that other streams also had groundwater inputs. Reaches were selected from among the watersheds to represent wide, narrow, and no buffer conditions.

Eight of ten buffered sites were established through CREP. Riparian buffer plantings in CREP exist in widths of 15 feet, 35 feet, and larger. A width of 15 feet (4.6 m) is the standard minimum for the Natural Resource Conservation Service (NRCS) hedgerow planting practice used by conservation districts that implement CREP buffers. A 35-foot (10.7 m) buffer width is the minimum width for the forested riparian buffer NRCS standard practice and is also the minimum width for which landowners are eligible for easement payments as part of the CREP program. Since plantings already exist in 15-feet (4.6 m) and 35-feet (10.7 m) widths, buffer width was used as a treatment. Sites were designated by buffer widths: 15 feet (4.6 m, n=7); 35 feet (10.7 m, n=3); and no buffer which had no planted woody vegetation (n=4).

Criteria used for selecting sites were that they had similar adjacent land uses and that the reaches were mostly channelized. Buffered sites were at least five years in age with mature, full canopies (Table 2). The area encompassing all sites had very similar agricultural land uses; the primary agricultural products of this region are blueberries, raspberries, and forages used in dairy production. All but one site was on land used for livestock feed production that received applications of liquid manure fertilizer or droppings from livestock actively grazing on site. The other site was adjacent to a blueberry field. Two sites were paired, meaning that they were adjacent reaches, a non-buffered site and buffered site, which could be either upstream or downstream. Buffer widths were measured perpendicularly from the edge of the waterway's wetted width to the edge of the rooted vegetation on each side of the stream. These measurements occurred during the first week of the study June 27-July 3, 2014. Actual buffer widths for two sites did not fall within the buffer width categories, but the sites exhibited enough similarity in vegetation shade quality to be included in one of the categories. The width for site VVA 15' measured 5 to 7' (1.5 to 2.1 m), but the vegetation stem and mean effective shade were similar enough to a 15' (4.6 m) buffer to be included in this width category. The same was true of site STB 0' where the bank was planted with shrubs, but the vegetation had not formed a significant canopy to shade the stream so it was included in the no buffer category for analysis purposes. All buffered sites had equal widths planted on both sides of the waterway except DAL 35', an east to west flowing stream, with the northern side planted 15' (4.6 m) wide and southern side planted 35' (10.7 m) wide. DAL 35' was included in the 35' buffer category.

Planting densities varied depending on the NRCS standard practice that considers the width of the waterway, the slope of the adjacent land, and habitat goals for the site. Hedgerow buffers (15’, 4.6 m) are typically one or two rows of woody species with a 4-foot spacing with at least two or three plant species. In contrast, riparian buffers in forested landscapes (35’, 10.7 m) usually have greater spacing and plant species diversity. Native lowland vegetation used in planted buffer sites were Oregon ash (*Fraxinus latifolia* Benth.), Pacific ninebark (*Physocarpus capitatus* Pursh), red osier dogwood (*Cornus stolonifera* Michx.), black twin berry (*Lonicera involucrata* Richardson), willow (*Salix* spp.), wild roses (*Rosa* spp.), Douglas spirea (*Spiraea douglasii* Hook.), snowberry (*Symphoricarpos albus L.*), and red alder (*Alnus rubra* Bong.).

b. Field Data Collection

Throughout the summer season, Hobo thermistors (Tidbit V2 Temp Logger, Onset Computer Corp., Bourne, MA) were used to measure air and stream temperature. Stream temperature was measured at the most upstream and downstream point at a site, so thermistors were approximately 100 m apart (Figure 3). The 100-m section selected for each site was not always at the beginning and end of a planted section. At times the 100-m site that was selected fell within the length of a longer planted section (DGR 15’, STA 15’, VPL 15’, and VSA 15’) because of adjacent land ownership. Water thermistors were shielded in hollow steel pipes with caps and were attached to t-posts. T-posts were driven into the streambed so that the thermistors were 5 cm above the substrate. Site air temperature was measured by one Hobo thermistor inside a plastic solar radiation shield (Pendant UA-002-08; and RS1 Solar Radiation Shield, Onset Computer Corp, Bourne, MA). Radiation shields were mounted to t-posts 1 meter above the ground, within the buffer understory, centered within the length of the buffer section. On one occasion, the air thermistor failed to continue monitoring, and temperature data were substituted from a nearby weather station maintained by Washington State University (AgWeathernet 2014). Field methods conformed to and continuous air and water temperature data were analyzed according to Washington State Department of Ecology (WADE) standard operating procedures (Ward 2011). For both 2014 and 2015, temperatures were monitored in 15-minute intervals from June 27 through August 21. To assure data quality, the calibrations of the thermistors were checked in an ice bath and at room temperature before deployment to ensure they operated within the manufacturers specifications (range -20 to 70 ˚C, accuracy ±0.21˚C from 0-50˚C). After deployment, thermistors were checked according to standard operating procedure by comparing the thermistors to a standardized thermometer (USEPA 2014). A determination of the consistent difference in degrees between a NIST thermometer (Control Company, Nazareth, PA, accuracy ±0.1˚C) and thermistors was made by comparing ten readings in multiple water baths. The water bath temperatures alternated between room temperature (~16.0˚C) and refrigerated temperature (~2.0˚C). I was unable to calculate drift since thermistors were only tested in this manner after at the end of the study.

Prior to placement of the thermistors, each stream channel was inspected for difference in temperature between the center of the channel and tile-drain input points along the substrate. At sites STA 15’, and VSA 15’ tile drains were visible above the water level of the channel. A thermistor with a 3 m cord (YSI 85, ±0.1˚C, YSI Inc., Yellow Springs, OH) was moved along the edge of the bank and stream bottom and monitored for change in temperature greater than 0.1˚C. When the tile drains may have been obscured by high flows within the channel, detection of their presence was based on the assumption that a difference in temperature would occur between the channel and incoming tile drain flow. A difference in temperature was not detected at any location during the pre-installation inspection. Since drain tiles became visible at some sites as water levels decreased, the lack of detection was likely due to the similarity in stream temperature and tile drainage temperature early in the summer. During base flow later in the summer stream levels were influenced mostly by groundwater. Little to no water was observed flowing from the tile drains during base flow conditions, so they did not affect stream temperatures.

Fish sampling was conducted the third week in July, 2014 and 2015. Samples were collected using a backpack electrofisher. A two-person team was used at each site; one operating the electrofisher (Appalachian Aquatics, Model AA-24, Morristown, TN) and the other capturing stunned fish with a hand net. At each site a 100-m long reach was divided into three sampling sections that were 30 m long with a gap between and each section received a single pass. Fish were identified to species, measured, and then released downstream from the point of capture as quickly as possible. No anesthetizing agent was used, the electrofisher was adjusted to the lowest effective voltage as possible, and the fish were handled quickly in the shade while data collection occurred (NMFS 2000). Relative abundance of fish is expressed as the number of fish species per unit of effort (or relative catch per unit effort CPUE; McCormick and Hughes 2001). The channels were homogenous with no distinct pool-riffle morphometry. The assumption for comparing relative abundances that were normalized by CPUE are that the rate of catch is proportional to the size of the fish population.

Physical site conditions were recorded to to identify relationships to water temperatures within each study site and included percent effective shade, average thalweg depth, flow, aspect, and air temperature (Mayer 2012). I did not discover any point sources of groundwater at sites in this study

Mean percent effective shade along each 100-m site reach was quantified through hemispherical camera photos according to Washington State Department of Ecology standard operating procedures (Stohr and Bilhimer 2008). Effective Shade is defined as “the fraction of total possible solar radiation that is blocked from reaching the stream surface and summed over a full day” (Stohr 2008). I used a digital camera (Cannon Eos Rebel xs, Cannon U.S.A. Inc.) fitted with a 4.5 mm circular fisheye lens (F2.8 EX DC, Sigma Corporation, Japan), which was attached to a hemispherical photography tripod mount system, as recommended by Stohr and Bilhimer (2008). Photos were taken mid-stream, 1 m above the water surface 10 meters apart along the length each reach so that each portion of the reach was represented equally. Percent effective shade was calculated for each photo using Hemi View © software (V. 2.0 , Delta-T Devices, Cambridge, UK), according to Washington Department of Ecology standard operating procedure (Stohr 2008). Hemi View software accounts for stream orientation, latitude, solar path angle, and day length. Sites were defined as having a latitude of 48.7475° N, and longitude of 122.485° W, with declination correction of the compass in the field of 16˚ 29’ 52’’ east of north. Percent effective shade was calculated for solar path and day length corresponding to August 1st at all sites. The date August 1st also corresponds with the time of the year when peak air temperatures are typically observed in the study site region (NOAA 2015). Hemi View software classifies each pixel in the digital photo as black or white. The threshold setting to classify the image was chosen manually for each photo to select the most appropriate representation. The number of black pixels is compared to the total number of pixels to calculate the percent effective shade. The percent shade for each photo within sites was averaged to calculate the mean percent effective shade for the site reach.

Vegetation along stream reaches was characterized according to number, species, height, and density. The 15 transects per side of the buffers were longitudinally equidistant 1 and perpendicular to the waterway channel (Figure 3). Data were collected for plants rooted within 1 m on either side of a transect. Typically, overhead canopy shade density is measured for tree species, but the density of vegetation beside the channel was also considered an important variable for the ability of the buffer to provide adequate shade at oblique solar angles, so diameter at breast height (DBH; 1.3 m from the ground) was recorded for all stems within the transect boundary zone. When stems were branched below 1.3 m, the DBH for each stem was measured at the 1.3-m height and then summed to represent the total DBH for that plant.

In June of both years, stream habitat and substrate quality were assessed, and water flows were measured. Three transects per stream were set perpendicular to the stream flow direction at the start, middle, and end of the reach. Each transect was divided into subsections of equal width no more than 30.5 cm wide. Velocity measurements were taken using a portable flowmeter (Model 2000, Marsh-McBirney, Inc., Frederick, Maryland) at each subsection at a point 0.4 X depth from the bottom where the depth at that point was less than 76.2 cm deep, following Rantz (1982). Flows for each subsection were calculated by multiplying width times depth times velocity for each cell, and cell flows were then summed to arrive at total discharge (Q).

Mean stream depth, thalweg depth, and stream width were calculated from measurements taken at the same transect locations used to take stream velocity measurements. Thalweg was the deepest point in a cross-section of the stream. The presence of a notable thalweg can indicate whether pockets of deeper cool water may be present in an otherwise shallow stream. To determine the most frequently observed (dominant) substrate type, particle size class, and texture, five independent observations occurred along each of ten transects within the stream. The size classes and methods were defined by a modified Wentworth scale (Bain 1999), and dominant substrate type was determined using the methods of Cummins (1962).

Nutrient concentration was assessed at each site from August 2014 through October 2015 through monthly “grab” samples at the downstream sensor location at each site to measure the input from sources throughout the watershed. Samples for nutrient levels (ammonia, nitrate, total phosphorus, and soluble reactive phosphorus) at each site were collected using methods for water sampling (IWS SOP # 22 2014) and tested following Western Washington University’s Institute for Watershed Studies Standard Operating Procedures (IWS SOP #6 2012). Monthly grab samples were collected in acid-washed polypropelene bottles at the downstream sensor location at each site. Once obtained, samples were placed on ice, filtered within 8 hours of collection, stored at 0˚ C for less than a month, and then analyzed in a Flow Injection Chemistry Analyzer (Flow Solution 3100, O I Analytical/ Xylem Inc., College Station, TX).

c. Data Analysis

Data from each type of measurement of physical stream and vegetation characteristics were analyzed separately using the statistical program R version 3.2.4 (R core team 2016). A comparison between water discharge data was conducted using analysis of variance (ANOVA) and the assumption of homogeneity of variance was verified using Levene's test. Assumption of Normality was tested using Shapiro-Wilk and led to data transformation by using the 8th root. Comparisons of shade levels between site width categories were tested using a non-parametric Kruskal-Wallis test.

Change in water temperatures for all measurements was examined but no patterns emerged from sites within width groupings. Change in water temperatures from upstream to downstream was calculated as the difference in "biologically significant heat units" (BSHUs). BSHUs are the positive difference between the mean daily temperature and 17.5˚ C, at a single thermistor location. The base temperature 17.5˚C is the USEPA criterion for the 7-day mean of the daily maximum water temperatures of salmon rearing and migration habitat that represents the upper limit for no adverse effects on fish health (WAC$_b$173-201A-200). At temperatures between 18 and 20˚C juvenile coho salmon growth stops (Stein et al. 1972, Bell 1973, Armour 1991) and an inverse linear relationship in abundance occurs between temperatures of 17 and 21˚C, where at 21˚C coho juveniles avoid the stream entirely (Frissel 1992). Thus the occurrence of BSHUs indicates that adverse effects on the health of salmon and trout fry and fingerlings will occur and that the magnitude of the adverse effects increases as BSHUs increase.

BSHUs were calculated based on temperature readings that were taken every 15 minutes, totaling 96 each day. From each reading, 17.5℃ was subtracted. The positive values were summed and then divided by 96 to provide a daily BSHU value for each thermistor location. (Equation1). The daily BSHU at the upstream thermistor location was subtracted from the BSHU at the downstream location to calculate the change in BSHU over the reach that occurred over each day (Equation 2). The mean of the differences between downstream and upstream BSHUs at each site was calculated for each week and reported over a total of eight weeks, 27 June to 21 August in 2015.

Equation 1. Daily BSHU = $\frac{\sum positive\ values[(T_1-17.5)+(T_2-17.5)+\cdots.(T_{96}-17.5)]}{96}$

Equation 2. Daily change in BSHU in a reach = (Day1 downstream BSHU) – (Day 1 upstream BSHU)

A smaller sample size than expected was the result of excluding the nine sites that never reached temperatures above the base temperature of 17.5 ˚C. In 2014 only one site had temperatures above17.5˚C. In 2015, sites that did not go above the base temperature of 17.5˚ C at either the upstream or downstream thermistor locations were: SSM 35’, STM 35’, SSY 15’, STA 15’, VPL 15’. STB 0’, VVB 0’, VVA 15’, VSA 15’ (Table 2). The remaining five sites were used to calculate BSHU statistics.

Because of the small sample size in the BSHU analysis, the relationship between air temperature and water temperature was explored to explain whether the microclimate effect created by the buffers was influencing water temperature. Maximum daily air temperatures “local” to the site locations were related to “outside” the site locations using a function that calculated Kendall’s tau in R 3.3.1 (R core team 2016). Kendall’s tau is a non-parametric coefficient of correlation. The “outside” air temperature readings were taken from a Washington State University temperature monitoring station at the Tenmile location (Agweathernet 2014). The “local” air temperature data were taken from temperature sensors deployed at each site, described previously. Similarly, local maximum daily air temperature was related to the downstream water temperature at its maximum daily reading over the entire study period June 27-August 21 in both years.

Further analyses were conducted to determine whether the shape of the channel was affecting the heat accumulation. The mean stream width and width to depth ratio

$$\text{Width to depth ratio} = \text{mean of } \left(\frac{\text{mean width; n=3}}{\text{mean depth of thalweg; n=3}}\right)$$

of each site was calculated. The week at all sites with the highest mean maximum temperatures was selected for comparison. A linear model of each day’s change in temperature during the period 6 A.M. to 5 P.M. was fitted to the points to determine a slope for each day. The stream width and width to depth ratio for each stream was plotted against the slope of daily temperature change and then evaluated for any relationship using Kendall’s tau in 2015. The same was done for 2014 except for the period 6 a.m. to 6 p.m. The end time for the daily time period over which the slope was calculated was chosen by identifying the time at which the maximum daily temperature occurred at all sites for each day of the designated week. The median time from these 98 observations (7days X 14 sites) was selected.

Fish abundance was calculated as catch per unit effort using longitudinal stream distance as the effort. Fish composition and abundance at each site were compared using multi-dimensional scaling (MDS) to detect clusters of similarity within each of the buffer width categories. The MDS analysis shows figures that represent the distribution of fish assemblages within each site. MDS is a non-parametric strategy that allows biological community data that is n-dimensional (multiple species in this study) to be represented in two dimensions (2-D). The distance of separation between assemblages at each site was calculated using Bray-Curtis ordination. This process ranks the species of fish in each site's population according to number. The goodness of fit between the 2-D representation and the predicted values from n-dimensional space are tested iteratively, and then moved graphically to minimize the difference. There is no scale for the representation of points. Interpretation of the 2-D figure lies in the relative distances between the points. Similar fish assemblages are located closer to one another in space than differing assemblages. The final difference in agreement between the observed distribution and the spatial representation of sites is represented as "Stress," which ranges from 0 to 1. At values greater than 0.15 the representation is considered suspicious, while values between 0.05 and 0.14 are considered a "good fit" (Kwak and Peterson 2007). Significance of dissimilarity was computed between treatment groups. The "R" value is the level of agreement between the observed population distribution and a theoretical test distribution, and it ranges from 0 to 1. The p-value is the measure of significance of the similarity between populations. The MDS Analysis was calculated in the statistical program "R" version 3.2.4 (R Core Team 2016) with the Vegan package (Oksanen et al. 2016). Fish samples were analyzed separately in 2014 and 2015. An exploratory analysis was used to test whether the sites could be clustered by buffer width according to similarity of cool-water or warm-water species. Data were transformed to represent presence-absence of these species at each site and an MDS was performed using the meta-MDS command in the package Vegan in R software (Oksanen et al. 2013).

Results

a. Air and Stream Temperature

Patterns in weekly air temperatures during the study period June 27-August 21 were similar in 2014 (Figure 4) and 2015 (Figure 5). In 2015, weeks 3, 4, 5 received 0.66 to 1.50 cm of rain and were much drier than 2014 (Table 3). These same weeks also had the greatest decrease in water temperatures at DAL 35’ (Figure 5). In 2014 weeks two, beginning July 4, and three, beginning July 11 had the warmest maximum temperatures. During these two weeks the highest temperatures recorded were 18.7˚C and 20.4˚C respectively. In 2015, weeks one, beginning June 27 and two, beginning July 4, had the warmest maximum temperatures. During these two weeks of 2015, 22.2˚C and 19.9˚C were the highest temperature s recorded at any site.

Differences in BSHUs were calculated for 2015 but not 2014. In 2014 when water temperatures were typically cool, only two sites exceeded the 17.5˚C base temperature, so the BSHU differences were not calculated. Changes in the 2015 mean weekly BSHU difference in downstream and upstream water temperatures through the summer are shown for sites DAL 35’, DEG 15’, ELA 15’, ELB 0’, and VSB 0’ (Figure 6). These were the only sites in 2015 that had temperatures above the base temperature of 17.5˚C. Positive differences in BSHUs indicate weeks where the downstream heat unit quantity was greater than the amount of heat units that were present upstream. Hence, biologically relevant warming at a site with respect to fish occurred over the 100-m length of a reach at temperatures above 17.5˚C.

The number of weeks where the mean BSHU difference was positive from upstream to downstream was fewer at the 35’ (DAL=2 weeks) site than the 0’ (ELB=7 weeks, VSB=1 week) and 15’ (ELA=6 weeks, DEG=4 weeks) sites. At the 35’ buffer, 25% of the eight weeks had positive BSHU and at the 0’ and 15’ sites more than 50% of the weeks had positive BSHU differences. The mean positive BSHU difference did not differ between the two 15’ sites by week (Levene’s test p=0.6, DF=7, F=0.78). BSHU differed among the 0’ sites week to week (Levene’s test p<0.05, DF=7, F=2.93). Site VVB 0’ had only one week where the temperature was recorded above 17.5°C and lacked variance, further reducing the number of sites that could be statistically compared among buffer widths.

Within the weeks with a positive BSHU difference from upstream to downstream the percent of the days that warmed was consistently higher at sites in the 0’ and 15’ groups than those in the 35’ buffer site after July 11, 2015 (Week 2; Table 4.).

When temperatures reached 17.5˚C in a waterway, the DAL 35’ buffer responded less to warming than the 0’ and 15’ buffers. In contrast, the DAL 35’ buffer site lost BSHUs from upstream to downstream. During weeks 3-6 BSHU temperature-equivalent values were between 0.03 and 0.09˚C lower upon exiting the buffer. The extent of the warming from upstream to downstream differed among the 0’ and 15’ site buffer widths. At water temperatures above 17.5˚C, during weeks 3-6, the BSHU values at 0’ sites had a positive accumulation of heat that equated to between 0.01 and 0.03˚C. At the 15’ sites BSHU values also had a positive accumulation of up to 0.04 ˚C.

The cumulative amount of time each site spent at temperatures above 17.5 ˚C was variable and not consistent within width categories (Figure 7.)

Relationships between air and stream temperatures

In 2014 and 2015, outside air temperature and local air temperature within each site were all significantly and strongly correlated (Table. 5). Local air temperature and downstream water temperature were also significantly correlated although there were variations in correlation strength among site widths (Table 6). In 2014, during the warmest week, July 11-17, and for the warmest week of the study in 2015, July 4-10, 2015, the mean daily rate of stream warming was compared to the mean width to depth ratio of the stream and no relationship was present (Figure 8., 2014; Kendall's Tau=0.09, p-value=0.667, Figure 9., 2015; Kendall's Tau=0.289, p-value=0.291).

b. Fish assemblage

The number of fish sampled in both years from all sites was similar. Total number captured across sites in 2014 was 618 and 2015 was 637 (Table 7). Warm water and native cold water fish species were found at all buffered and non-buffered sites (Table 7). Three fish species, three-spined stickleback (*Gasterosteus aculeatus*, L.), coho salmon (*Oncorhynchus kisutch*, Walbaum), and cutthroat trout (*Oncorhynchus clarkii*, Dymond) represented 70% or more of the total number captured in each year (Table 8.). Other species detected in the surveys were non-native species; bluegill (*Lepomis macrochirus*, Rafinesque), pumpkinseed (*Lepomis gibbosus*, L.), and brown bullhead (*Ameiurus nebulosus*, Lesueur) and native species; largescale sucker (*Catostomus macrocheilus*, Girard), sculpin spp. (*Cotus spp*.), Chinook salmon (*Oncorhynchus tshawytscha*, Walbaum), rainbow trout (*Oncorhynchus mykiss*, Walbaum), and lamprey spp. (*Lampetra richardsonii*, Vladykov & Follett and *Entosphenus tridentatus*, Richardson; ammocetes were not distinguished).

No pattern in fish assemblages related to site buffer widths was observed in the MDS analysis in 2014 (R=0.06 p=0.29 stress=0.1) or 2015 (R=0.006, p=0.46, stress=0.1). Sites within buffer-width groups did not tend to form clusters of points that were distinguishable from other clusters (Figure 10). No significant clusters in MDS analysis were found when sites were tested for similarities by cold-water or warm-water species (stress=0.14, R=-0.02, p=0.53)

c. Shade and Vegetation

Vegetation at the 35’ buffered sites had lower species richness and was less dense with larger DBH and taller plants than in the 15’ sites. Planting density in the 15-foot buffered sites was approximately three times greater than in the 35’ sites (Table 9). The dominant plant species in terms of frequency of occurrence in five of the seven 15’ buffered sites was wild rose (*Rosa* spp.). Douglas fir (*Pseudotsuga menziesii*, Mirb.) and Oregon ash (*Fraxinus latifolia*, Benth.) dominated the other two 15’ buffer sites. Bank vegetation at 0’ buffer sites was dense stands of reed canary grass (*Phalaris arundinacea*, L.*),* often with thickets of Himalayan blackberry (*Rubus armeniacus*, Focke) and invasive herbaceous weeds. The height of the grass and deeply incised channels account for any shade measured at these sites. Animal damage reduced native plant density in the VSA 15’ buffered sites, and this area was overgrown with Himalayan blackberry, which contributed to some of the shade over the waterway.

Effective shade was similar between 15’ and 35’ buffers. Maximum stream surface shade levels at the 35’ and 15’ buffer sites reached 97% in certain parts of the buffered 100-m reaches in both years. Minimum shade levels were as low as 34.4% at parts of the 15’ buffers (2015) and 43.4% in 35’ buffers (2015). Mean effective shade for buffered sites ranged between 74.8% and 85% with similar standard deviation (Table 9 and Figure 11).

d. Physical stream characteristics

Dominant substrate at all but one site was fine material ranging from peat to sand in 2014 and 2015; one site had coarse gravel (Table 10). No significant difference in stream flow among site buffer widths occurred within years (2014: ANOVA, p=0.13, F=2.1; 2015: p=0.25, F=1.44, alpha=0.05). Flows were significantly lower at 12 of the 14 sites in 2015 than in 2014 (Figure 12, Table10). Two exceptions, sites SSY 15’and SSM 35’, are adjacent to each other on the same stream, Crystal Springs Creek, which could have caused them to have different flow patterns than the rest of the creeks in the watershed. In some cases flows at more than three transects were measured per site. To create even sample sizes for water flow comparisons, three measurements per year were randomly selected when more than three existed.

e. Nutrient concentrations

Fluctuations in nutrient concentrations, Total Phosphorus (TP), Soluble Reactive Phosphorus (SRP), Total Nitrogen (TN), Nitrate, and the ratio of TN:TP, showed seasonal patterns at all sites (Figures 13-17). The lowest nutrient concentrations were observed during spring and summer months: May, June, July, and August. The highest were observed in late fall and winter. Mean total phosphorus was below 50 ug/L during most months (Figure 13). Total nitrogen levels were below 2000 ug/L, the EPA’s clean drinking water standard, at 50% of the sites on 80% of the sample dates.

Total phosphorus differed among all three buffer width categories but not significantly (Figure 18). The lowest mean concentration of total phosphorus across all sample dates (29.1±5.49 ug/L) occurred in the 35’ buffer width category. However, the mean concentration of the 15’ buffer group (49.2±21.7 ug/L) was higher than the 0’ buffer group (36.8±7.74 ug/L). For soluble reactive phosphorus the mean concentrations across sampling dates at the 35’ and the 0’ sites were similar (5.52±2.14 ug/L, 7.11±0.840 ug/L), and both were lower than the 15’ category (14.90±14.78 ug/L).

Discussion

Stream temperature and shade

My expectation for this study was that water temperatures would be primarilly driven by shade and the micro climate shade creates over the water surface. I expected water temperature to fluctuate in a similar fashion when buffers, regardless of their width, could provide similar effective shade levels. I expected this because other factors such as tree height and leaf area index can influence maximum stream temperatures more than buffer width (Sridhar et al. 2004, Dewalle 2010). However, my results did not support the literature since the 15’ and 35’ buffer widths provided similar levels of effective shade but differences in BSHUs reflected temperature loss for the 35’ site and some loss and some gain for the 15’ sites during times of the day when maximum temperatures occurred (Brown et al. 2010, Ryan et al. 2013).

The great variation in the weekly BSHUs within each site category confounds the study’s ability to identify the effectiveness of narrow buffers. The variation in effectiveness in water temperature maintenance between the 35’ and 15’ buffers could have been caused by factors other than buffer width. Since flow, substrate, and effective shade were similar among all width categories, possible influences on water temperature were groundwater influx (Harper-Smith 2008), inputs from tile drains, removal of water from the channel for irrigation, and other forms of overhanging vegetation closer to the stream surface such as reed canary grass. The length of the site reaches could have limited the ability to detect a change in water temperature as well. All of these factors control aspects of a stream's thermal sensitivity. Thermal sensitivity, in turn, controls the extent to which temperature change can be detected over a set length of stream.

Some factors such as ground water inputs were not evident based on lack of change in flow and temperature so the in influence was likely negligible, but others, including withdrawal for irrigation, were noted. Compared to shade and air temperature, even over reaches of 500 m, groundwater has been found to have much less of an impact on cooling of downstream temperatures (Harper-Smith 2008). Prior to this study I had no information as to whether the creeks in my study were used for irrigation. Especially warm, dry weather in summer, 2015 necessitated irrigation of agricultural fields adjacent to the study sites for many weeks of the study period. I documented irrigation withdrawals at VSB 0' where the out-take pump was located at the upstream thermistor site and withdrawals also likely occurred upstream of sensors at DAL 35', STB 15', SSM 35', and SSY 15'. Reduction of water in the channel may have increased the thermal sensitivity of the stream because lower volume and lower velocity increase the rate of heat exchange and could make the effect of buffer width more detectable. However, the potential for increased heat exchange may have been offset by short, herbaceous vegetation at the site. Reed canary grass was present at site

VSB 15’ on the bank and in the stream covering the narrow channel, so shading by grass may have reduced the exposure to direct solar warming at this site.

Reach lengths of my study sites may also have affected my ability to detect changes in temperature. In this study, a 100-m reach may not have been a long enough distance to detect differences in temperature from beginning to end of the reach. Most other studies linking water temperature change to riparian vegetation used reach lengths ranging from 100 to 2,000 m (Barton et al. 1985; 0.1-3km, Harper-Smith 2008, Ryan et al.2013; 300m). Planted buffers in general tend to be short in this area, and those longer than 100 m are often on separate parcels of land. Because it is difficult to secure the permission of multiple landowners, my selection of study sites was limited.

Wide variability in water temperature change could be from physical factors of the stream that contribute to high thermal sensitivity, the rate at which water temperature changes. The rate of thermal change is a factor of intensity of solar radiation, current velocity and volume, and cross-sectional shape of the stream, among others. Furthermore, precipitation, or groundwater input s impact the thermal sensitivity, but, the magnitude of that impact is reliant on the previous factors mentioned. For example, a shallow, wide stream with little shade and bedrock substrate will warm considerably faster (high thermal sensitivity) than a deep, narrow, shaded stream with cobbles and gravel (low thermal sensitivity).

The one site in the 35 foot buffer width category that was analyzed for BSHUs, DAL 35’, had the greatest variation within weekly BSHUs possibly because of high thermal sensitivity. At the DAL 35’ site, the water level is shallow (mean thalweg=0.17 m) while the width of the creek is wide (mean width=2.2 m), making the thermal sensitivity of the stream very high. Typically, a rain event can lower the temperature of a creek in a short period of time. Groundwater input can also affect the thermal sensitivity of the stream by providing both an influx of cooler water and increasing the volume present in a reach. Without buffering by cool water input, stream temperature would change more rapidly. The presence of groundwater sources at DAL 35’ could explain why BSHU differences decreased at this site when compared to the 15’ buffer category sites where air temperatures were similar. DAL 35’ had the most rapid response to change in air temperature in comparisons of the average rate of temperature increase across sites during the warmest week of the season (Week 2: 7/4/15-7/10/15); therefore, it showed the greatest thermal sensitivity during this period of maximum warmth. Precipitation also may have affected the variability of the 35’ buffer site BSHU differences. Weeks 3, 4, and 5, received the greatest amounts of rainfall during the study season and were the same weeks that had the greatest levels of variability in BSHUs (Table 3, Figure 6). During this period of variability in temperature, the stream was experiencing decreases in BSHUs from upstream to downstream. I observed up to a 2.3°C loss in BSHUs. The pattern at DAL 35’ could be applicable to other streams in the Nooksack drainage with groundwater input and high thermal sensitivity.

Fish assemblage

I predicted that fish assemblages would show a response to differences in environmental conditions caused by differences in buffer widths, but I did not find any relation between fish and buffer width categories. This suggests that buffer width did not affect the composition of fish assemblages. If this finding was a false negative, a factor within the analysis that could have compounded my ability to find differences, if they were present, were the low number of species found overall and the relative rarity of some species. The sensitivity of the Bray-Curtis ordination that I used with MDS has been shown to be limited by species having very low or very high abundance (Kwak and Peterson 2007). The three-spined stickleback was in high abundance in this study, but removing it from the MDS analysis did not change the results. So I posit the lack of detection of differences in assemblages was outside the analysis process and lies in aspects of the field study.

Fish are highly responsive to changes in physical habitat. Differences in fish abundance among buffer width categories could not be consistently attributed to single factor that affect the presence of more sensitive species. Three main factors present at the sites in this study are, persistently degraded landscape surrounding all the sites, variation of habitat structure complexity within buffer width categories, and lack of an adequate amount of time to measure change in relative abundance due to water temperature changes from buffer installation (Wichert and Rapport 1998.)

Channelization and dredging are the mark of persistently degraded conditions in the agricultural region containing my study sites. Lack of channel structure and complexity, which was absent from nearly all sites, could have been the driving force in fish assemblage make-up. Neither the 15’ or 35’ buffers were contributing large woody debris to the channel based on my observations, or, if they were, it was removed to promote drainage. Streams in this study were channelized and most lacked in-stream habitat characteristics that create conducive conditions for fish such as larger substrate and structural features that form pools and riffles. Habitat was highly homogenous.

Although the relative total abundance of fish at each site did not differ, the relative abundance of individual species differed. A prime example of this is juvenile coho salmon mean relative abundance was approximately 7-times greater in sites with either a 15’ or 35’ buffer than those with no buffer (Table 8). This pattern was observed in both the 2014 and 2015 surveys. Coho presence within buffered sites gives evidence of buffer effectiveness for habitat since stream conditions must be maintained for 1 to 2 years while juveniles rear in these agricultural waterways. Other species of salmon and trout leave rearing streams in less than one year.

Variation of habitat quality, as defined by abundance and diversity of fishes, within buffer width treatments did exist, but not enough to influence the trend of the group. Sites with the greatest relative abundance of coho salmon were the anomalous sites with habitat features. The sites SSY 15’ and SSM 35’ were improved by the Regional Fisheries Enhancement Group, Nooksack Salmon Enhancement Association (NSEA). NSEA added large root wads, log braces, and boulders in addition to planting vegetation. These two sites were also influenced by lower initial water temperatures from cool water inputs of a spring upstream at both sites. STB 0’ had “islands” of reed canary grass growing on gravel deposits mid-channel and some emergent vegetation (*Elodea canadensis* Michx.) that provided cover to juvenile salmon and trout. DEG 15’ had thick mats of *Lemna minor* L. Dominant substrate and stream flow were similar between site widths so it is possible that in-stream vegetation and structure made a difference in maintaining better conditions for coho salmon than streams without.

The length of time during the day that water temperatures are very high matters; fish can survive short periods of thermal stress near lethal temperatures (Bjornn & Reiser 1991). In lowland, rain-driven stream reaches, such as our study sites, I expected to measure a difference in habitat use (i.e. fish relative abundance) between shaded and un-shaded sites because the buffered sites spent fewer minutes at temperatures above 17.5° C than sites without buffers (Figure 7).

Conclusion

Observations of individual 35’ and 15’ wide buffers suggest they may be more effective in reducing water temperature increases than no buffer during the warmest maximum temperatures of summer. However, other environmental variables and small numbers of sites within buffer width categories prevent a more specific statement about the relationship.

The relative abundance of fish was not related to buffer width categories. BSHU differences in 2015 at the 35’ site allowed for the stream to cool at above the base temperature of 17.5°C. So, at very high ambient air temperatures the 35’ buffer prevented the stream temperature from warming above temperatures potentially stressful to rearing and migrating salmon and trout. But, this site had no coho at all, further confirming the study’s inability to compensate for habitat and conditions which introduced variability into the analysis.

It is difficult to compare water temperature maintenance directly to other studies because of the way temperature maintenance was calculated in BSHUs rather than the mean 7-Day maximum daily values. But, as global climate change continues to cause increases in temperature at northern latitudes, perhaps this research gives an important look at how narrowly planted riparian buffers perform at maximum summer temperature conditions.

Future research on human-wildlife interfaces such as CREP buffer zones should be directed toward improving in-stream habitat conditions for salmon in addition to shade. Shading alone in narrow agricultural waterways may not effectively moderate stream warming to address the habitat needs of migratory salmon and rearing juveniles. Some research might be spent on investigating micro-habitats within narrow buffers. I observed small beaver dams at site VVA 15’ (Figure 19) and SSY 15’ that created pools. At site VVA 15’, the willow roots at the edge of the waterway created small (~15 cm^3) pockets of slow water and increased habitat complexity in otherwise channelized agricultural waterways. Investigating the effects of tile drainage may also provide insight on how better to provide salmon and trout thermal habitat. An unexpected observation in this study from both years at the same 0’ buffer site, was the presence of juvenile coho salmon at the cool outflow of a tile drainage pipe in an otherwise oxygen depleted environment. Experimenting with creative and effective ways to provide artificially constructed physical habitat while addressing farmer’s needs of irrigation, and drainage would be instructive to land managers at government agencies who implement habitat programs.

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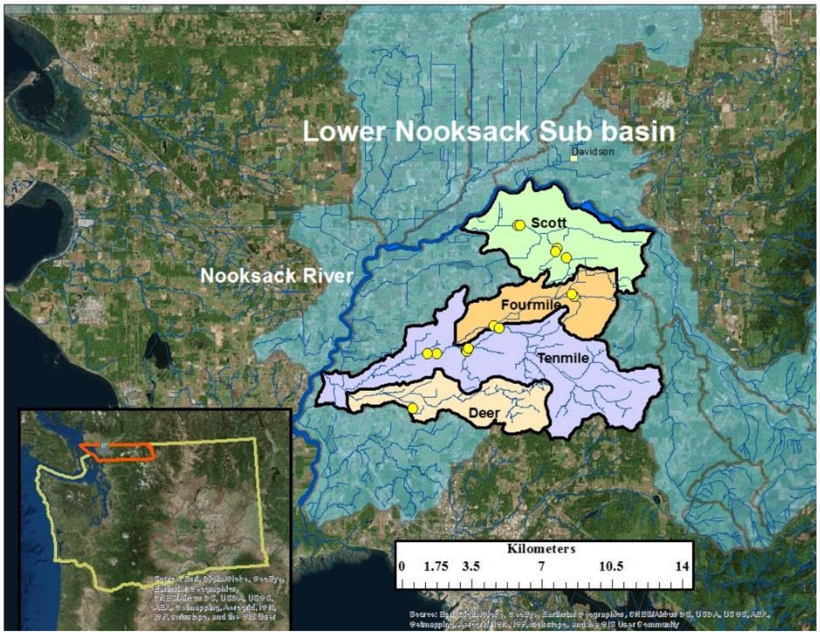
Table 1. Literature review of papers relating to the effect of buffer width on water temperature and fish populations. The studies are grouped by the type of land use at the study sites, timber harvest or agricultural production.

Citation	Vegetation Characteristics	Buffer Width (m)	Effective Shade (%)	Stream Width (m)	Water Temperature (C˚)	Fish population or relative abundance	Empirical Data or Model	Study Design/Results	Location
	Timber Land Buffers								
DeWalle 2010	30m tall	6-7 m-E-W 18-20 m N-S	80%	Stream to Height ratio = 5	NA	NA	Model	Model predicted potential solar radiation to stream center using Beer's Law. Assumption: buffers were on level terrain, no overhanging veg, no meandering, 40degree latitude, and even slope.	40˚ N lattitude
Zwieniecki and Newton 1999	timber lands (conifer)	8.6-30.5, mean 21.1	buffer zone: 78% Natural: 83%	ave: 3.4 (0.7-7.0 m)	prevent increase outside the normal warming trend for summer max temp	NA	Emperical	Measured change in weekly moving ave max stream temps at upstream and downstream edges of harvest unit (350-1600 m) and 150 and 300m downstdream in recovery zones.	western Oregon
Castelle et al. 1992/ Castelle et al. 1994 Same info	all four studies: timber lands (conifer)	15-60	85%	variable	maintain pre logging temperatures within 1 ˚C	NA	Model and Emperical	Wetland Buffer Review Appendix C: four studies presented on temp maintenance in summer in "Temperature Moderation" section	Pacific Northwest
Haberstock et al. 2000	Naturally vegetated/ 15 m tall red spruce, balsam fir	30	variable	variable	NA	NA/ Atlantic salmon habitat requirements	Literature Review	Developed a Buffer width Key based on slope, soil, % canopy closure at sites within watershed, water features. Key sets fixed width guidelines for salmon protection.	Maine
Wenger 1999	variable	10-30	NA	NA	maintain stream temperatures	NA	Model and Emperical	Literature Review	Georgia
Wilkerson et al. 2006	timber land (conifer) >15m tall after logging	11 m partial cut buffer	>60%	mean: 1.9-4.5	Pre-treatment 11.9-15.6 ˚C. Weekly max increased 1-1.4 ˚C, non sig.	NA	Emperical	Studied the weekly maximum average of daily stream temp. 400m downstream of harvested area and 100 m downstream in recovery zone.	Maine
Broadmeadow and Nisbet 2004	timber land	10 - 30	NA	NA	Maintain water temperatures	variable but generally positively associated with shade/veg	Emperical	Literature Review: Global Review of width recommendations. Buffer widths recommended to maintain water temperatures range between 10 and 30 m	Global
Moore et al. 2005	timber land	10 - 30	variable	variable	most streams increased 0.3-3.8 ˚C (either mean monthly max temp, mean weekly max, or weekly temp mean) after various logging treatments	NA	Emperical	Literature Review: Focus on studies with BACI design	Pacific Northwest
Degroot et al. 2007	timber land (conifer)	presence/ absence	presence/ absence	variable	Logging warmed the stream 1 ˚C	cutthroat trout abundance remained the same	Emperical	BACI design 6 yrs. Control streams sig > abundance than logged, but no difference between before and after logging on abundance. Secondary findings-average and max daily water temperature increased 1 degree after logging	Maple Ridge, British Columbia, CAN
Janisch et al. 2012	timber land (conifer ~40 m tall and red alder in riparian zone)	Continuous: 10-15 m, Patch Buffer: variable	Canopy and Topographic density ave: 94% before harvest. After; clearcut: ave 53%, Continuous: ave 86%, Patch Buffer: ave 76%	ave discharge: 0.3 L s^{-1}	All sites increased max daily temp; Clearcut: ave- 1.5 ˚C, Continuous: 1.1 ˚C, Patch Buffer: 0.6 ˚C	NA	Emperical	Over 7 years compared max daily temp in continuous buffer, patch buffer, clearcut; BACI design during July and Aug . % canopy cover not significant explanatory variable. Streams with gravel more thermally unresponsive. Streams with connected wetlands and longer surface flow distance above temp monitoring site were more responsive.	western, WA

Table 1 Continued.

Citation	Vegetation Characteristics	Buffer Width (m)	Effective Shade (%)	Stream Width (m)	Water Temperature (C˚)	Fish population or relative abundance	Empirical Data or Model	Study Design/Results	Location
	Agricultural Buffers								
Barton et al. 1985	natural vegetation agricultural watershed	10 m	80%	NA	≤22 ˚C =trout stream	presence/absence/ Occupancy- brook, brown, rainbow	Model	Measured % bank within 2.5 Km upstream. Influence of buffer declines after 1km upstream	Canada
Ryan et al. 2013	adjacent to agricultural grassland (banks without trees=unshaded) Buffer tree height 8-15 m	1-2 trees deep	presence/ absence	Large ≥8m Small ≤4m	Difference in temp. Down-Up were lower in shaded sections. Differentials were lower in large streams vs. small . Small shaded streams can cool up to 1 ˚C	Atlantic Salmon	Emperical	compared 300m long stream sections up - down temp. *stream substrate granite, shale, slate	Ireland
Blann et al. 2002	agricultural (62% land use) Variety: grazed grass, unmanaged wooded buffer, Successional-ungrazed area with grass, shrubs, forbs	not presented	(% Canopy Cover) Wooded- 49.2%; Successional-25%, Grazed-15.3%	Stream site 6-7 (model used 2.5 m to predict the amount of shade given by grazed, wood, Succ.)	(spring fed) Wooded temp change =0.1C/Km, SD 0.1; Grazed changed 0.4/Km, SD 1.4; Successional changed 0.1 C/Km, SD 0.8	brown, brook, rainbow trout	Model-	SNTEMP-predicts mean and max temps throughout stream length for distinct sections. Tested how these segments would respond to different vegetation types. Temperature only weakly correlated with % shade (r2=0.288, P=0.09). Mean temp change was not significant among buffer types.	Minnesota
Harper-Smith 2008			Model-100% canopy cover had stronger effect on max stream temp than groundwater input			NA	Model	SSTEMP model compared predicted temps to modeled temps. Used model to predict stream temps based on full shade cover and 50% groundwater inputs.	western Washington
Pess et al. 2002	agricultural and conifer	% forest cover within 100 m	>60%	NA	NA	annual spawning count-Coho positively correlated with higher density veg	Emperical	Study examined data from a 14yr period and compared fish population between years (local and watershed scale): positive corr. with % forest cover. Negative corr. with agricultural/urban land use.	Snohmish R., western Washington
Wichert and Rapport 1998	planted buffers in urban landuse	NA	> 85% vegetation bank cover	NA	NA	Fish Community IBI, improved	Emperical	Compared Fish community sensitivity to land use in agricultural and urban streams. Over time. Where vegetation increased, fish condition improved	Guelph, Ontario, CAN

Figure 1. Reed canary grass in VVB 0’ buffer site (un-planted) 2015.



Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community

Figure 2. Main Map: Study sites are marked by yellow circles within their respective drainages of the Lower Nooksack Sub basin; Scott, Fourmile, Tenmile, and Deer Creek. Lower Left Map: An outer extent showing the boundary of Washington State in yellow, Whatcom County in red, and Lower Nooksack Sub basin in blue.

Table 2. Study Site Characteristics.

a. Sites in the table are grouped by drainage location within the Lower Nooksack Sub-basin.

b. Paired buffers are those with a 0’ buffer reach adjacent to either a 15’ or 35’ buffered reach.

Site Code	Drainage	Waterway Name	Paired (Yes/No)	Riparian woody Width (ft)	Additional grass buffer (Yes/No)	Land Use	Flow Direction	Tile drainage	Semi-Natural or Channelized (N/C)	Date Planted/ buffer age (years)	Notes
DAL	Scott	Elder Ditch	N	35	N	berries	East - West	yes	C	2006	
ELA	Scott	Scott Ditch	Y	15	N	pasture silage/ crops	East - West	Yes	C	2009	South bank planted earlier than north.
ELB	Scott	Scott Ditch	Y	0	N	corn silage /crops	East - West	Yes	C	N/A	
VSB	Scott	Elder Ditch	Y	0	N	dairy pasture corn silage	South - North	yes	C	N/A	
VSA	Scott	Elder Ditch (Scott drainage)	Y	15	N	dairy corn silage	South - North	Yes	C	>5	Himalayan blackberries established in large quantities.
VVB	Fourmile	Fourmile Creek	Y	0	N	dairy corn silage	Northwest-Southeast	yes	C	N/A	
VVA	Fourmile	Fourmile Creek	Y	5-7	Y, 20'	dairy corn silage	Northeast-Southwest	Yes	C	>5	Included in the 15' category because of vegetation quality
STA	Fourmile	Fourmile	N	15	N	grass/ corn silage	East - West	Probable	C	>10	
VPL	Tenmile	Tenmile Creek	N	35	N	pasture	East - West	No	N	>10	Dammage to buffer from cows, no fence.
STM	Tenmile	Tenmile Creek	N	35	N	pasture	East - West	Probable	N	>10	
STB	Tenmile	Tenmile	N	0-5	N	grass/corn silage	East - West	Probable	C	<4	Included in the control category because of vegetation quality
SSM	Tenmile	Crystal Springs	N	15	N	pasture	South - North	No	N	>10	Mature overstory, almost no understory vegetation
SSY	Tenmile	Crystal Springs	N	15	N	pasture	South - North	no	N	>5	Overstory is young
DGR	Deer	Deer Creek	N	15	N	pasture	South - North	yes	C	>10	

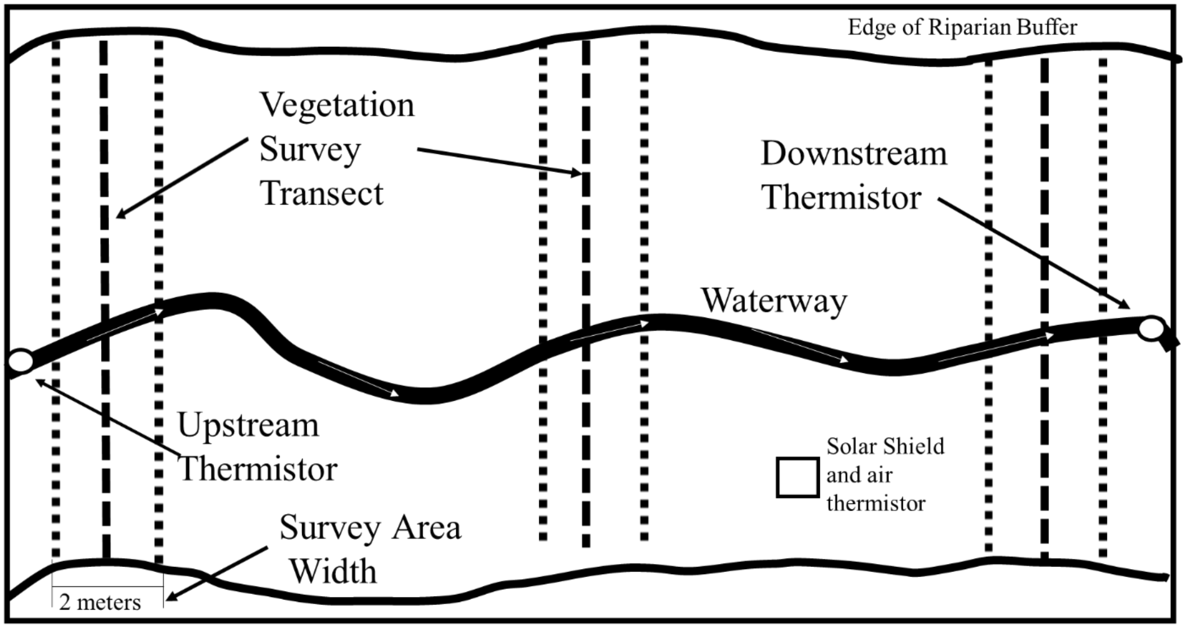


Figure 3. Illustration of data collection setup.

a. Fifteen vegetation transects were sampled on each side of the channel along the 100-m long sites.

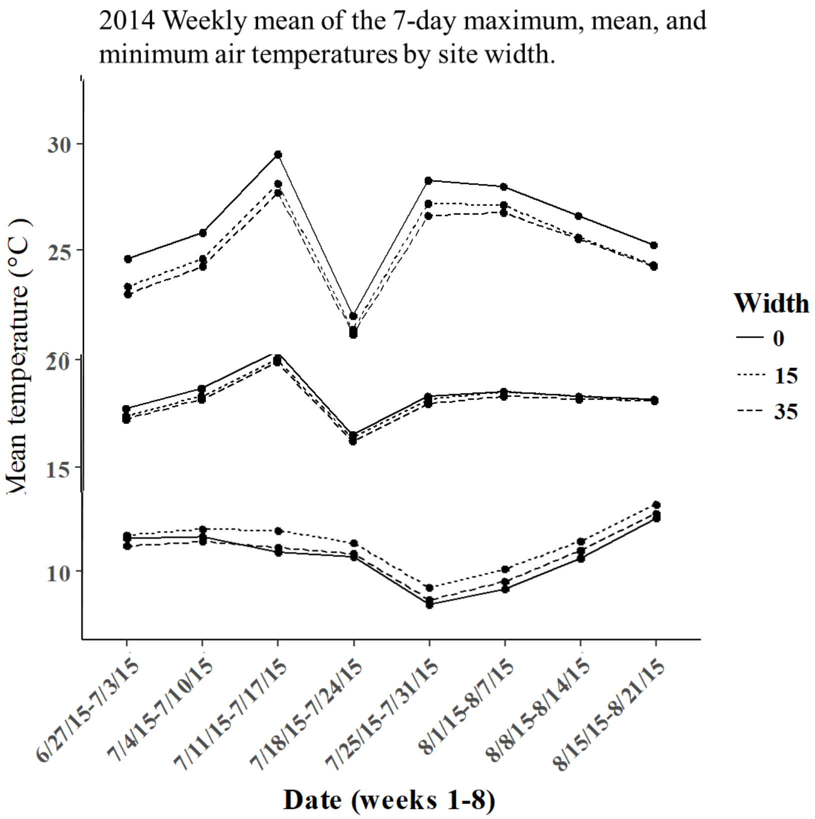


Figure 4. Mean, maximum, and minimum air temperatures by week and site width in 2014.

a. Data comes from hobo thermistors inside the buffer away from the waterway edge.

b.Daily means for each site were calculated from 15-minute measurement periods; n=96, weekly mean temperatures; n=7.

c.Sample size for weekly mean temperatures by site width: 0’=28, 15’=49, 35’=21.

d.Data is missing from site DAL (35’ width) for weeks 3-8.

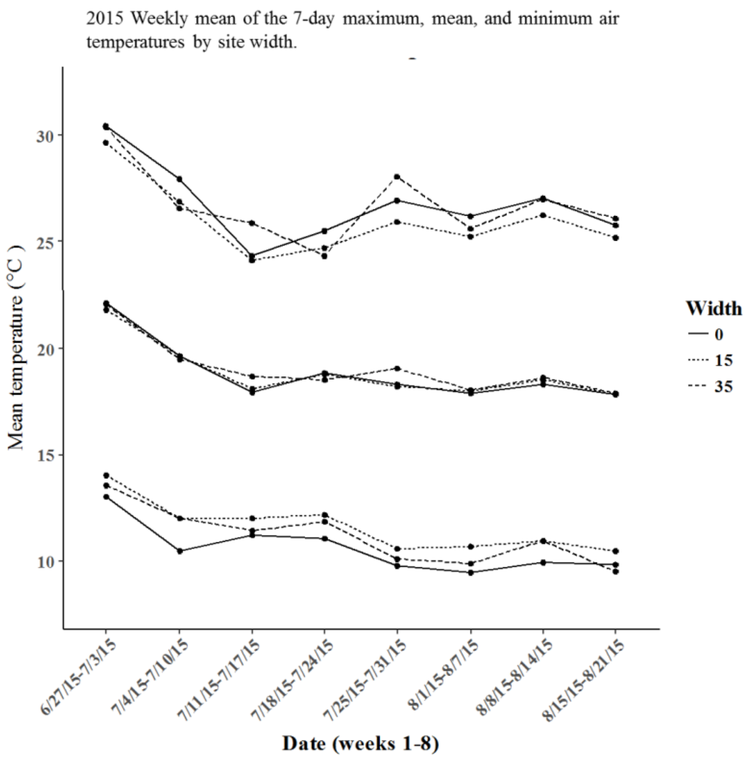


Figure 5. Mean, maximum, and minimum air temperatures by week and site width in 2015.

a. Data comes from hobo thermistors inside the buffer away from the waterway edge.

b.Daily means for each site were calculated from 15-minute measurement periods; n=96, weekly mean temperatures; n=7.

c.Sample size for weekly mean temperatures by site width: 0’=28, 15’=49, 35’=21.

d.Data is missing from site VVB (0’ width) for weeks 3-8.

Table 3. Precipitation in centimeters during the 2014 and 2015 field season by week.

a. Dates listed for each week correspond to the first day in the week and weeks are consecutive.

b. Data is from the Tenmile Ag weathernet station in Whatcom County, WA.

		Precipitation (cm)	
Week	Date	2014	2015
1	27-Jun	0.33	0
2	4-Jul	0	0
3	11-Jul	0	1.2
4	18-Jul	3.12	1.5
5	25-Jul	0	0.66
6	1-Aug	0	0.1
7	8-Aug	2.29	0.3
8	15-Aug	0.03	0
Total		5.77	3.76

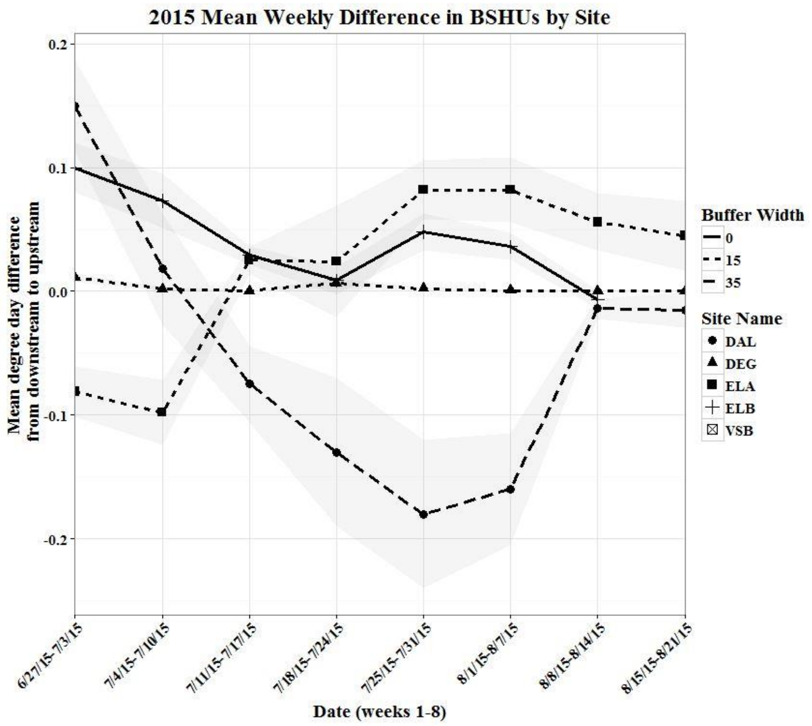


Figure 6. Mean weekly difference in downstream and upstream BSHUs by site and buffer width.

a. Buffer 0: n=14 (two sites; each with 7 days of measurements), Buffer 15’: n=14, Buffer 35’:n=7.

b. Sites with temperatures never reaching above 17.5˚C were removed from this analysis (SSM:35’, STM:35’; SSY:15’, STA:15’, VPL:15’; VSA: 15’; STB:0’; VVB: 0’).

c. Grey shadowed area shows the 95% confidence interval.

d. VSB 0’ buffer had one day during week 4 that went above 17.5. That BSHU difference is 0.001. This point is not visible on the figure.

Table 4. Percent of days that BSHU values were positive, indicating warming from upstream to downstream, for each week in 2015 by buffer width category.

	Week Date	27-Jun	4-Jul	11-Jul	18-Jul	25-Jul	1-Aug	8-Aug	15-Aug
	Week Number	1	2	3	4	5	6	7	8
Buffer Width[a]	0'	50[b]	50	43	36	43	36	14	7
	15'	36	7	43	36	50	36	36	29
	35'	86	57	0	0	0	0	0	0

a.Weekly % of days exceeded by 17.5°C= $\frac{\sum[\#\ d\ siteA+\#d\ site\ B+\#d\ site\ C]}{(\#sites*7days)}*100$

where d=number of days in a given week that the difference in BSHUs was positive

b. 0’n=2 sites; 15’ n=2 sites; 35’ n=1 site

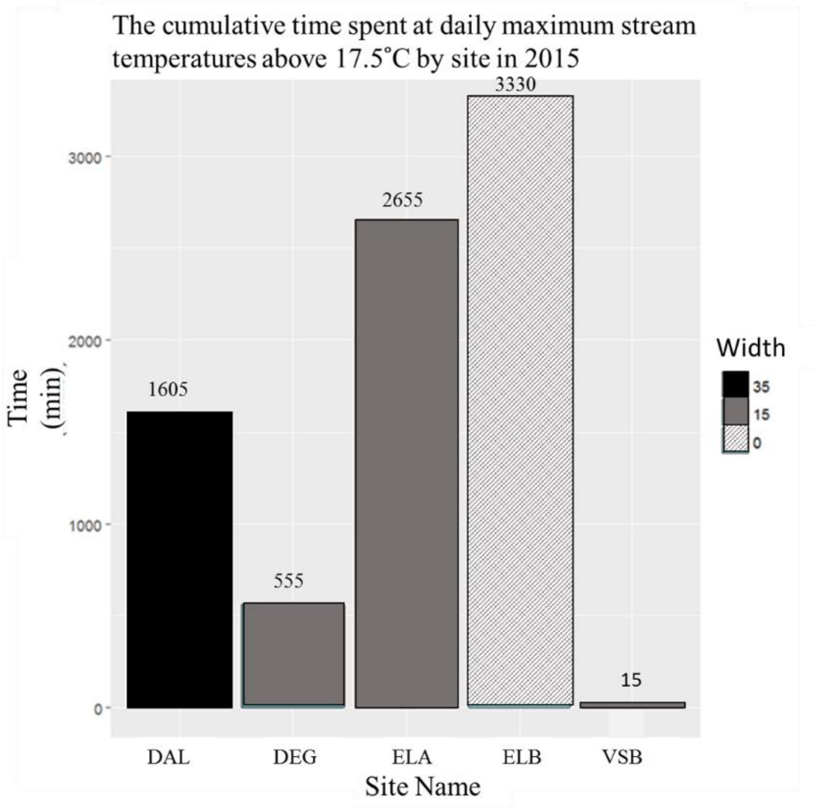
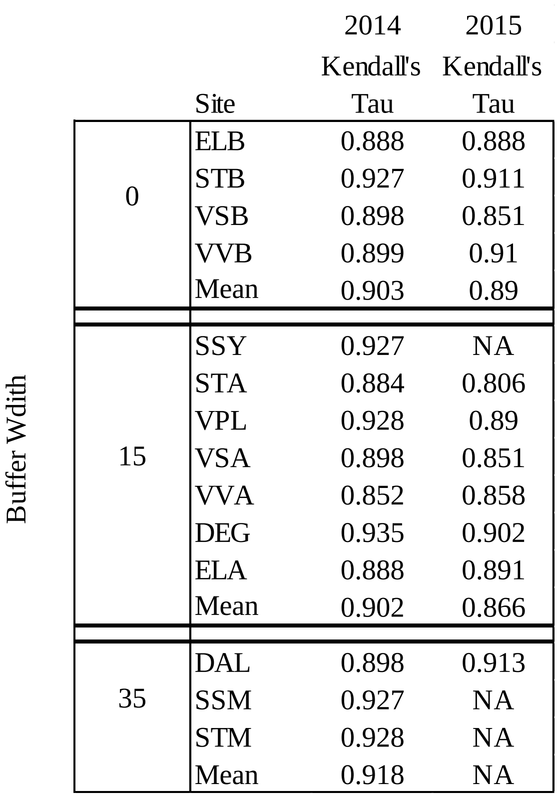


Figure 7. Cumulative time spent at the daily maximum stream temperature above 17.5℃ for the same sites in 2015 that were analyzed for BSHU differences.

Table 5. and 6. Individual results by site and mean values of results by buffer width treatment of ranked correlations between reference air temperature (Ag weathernet station) and local air temperature near the sites (Table 5.) and local air temperature and downstream water temperatures (Table 6.)

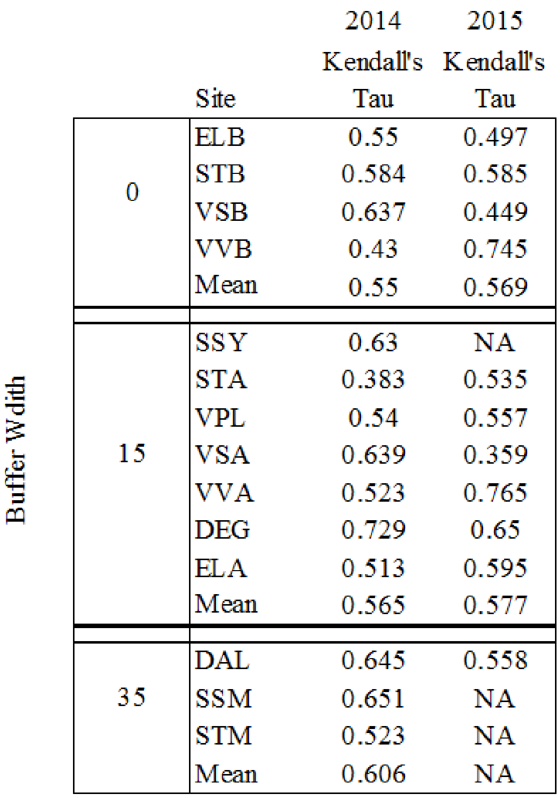
a. Kendall’s Tau is the correlation coefficient for all 15-minute time period temperature measurements recorded from June 27-August 21 in 2014 and 2015.
b. All correlations were significant at the 0.05 alpha level.
c. NAs signify missing data at the near site air temperature sensor in 2015.

Table 5. Reference Air Temperature VS. Local Air Temperature



Buffer Wdith		Site	2014 Kendall's Tau	2015 Kendall's Tau
	0	ELB	0.888	0.888
		STB	0.927	0.911
		VSB	0.898	0.851
		VVB	0.899	0.91
		Mean	0.903	0.89
	15	SSY	0.927	NA
		STA	0.884	0.806
		VPL	0.928	0.89
		VSA	0.898	0.851
		VVA	0.852	0.858
		DEG	0.935	0.902
		ELA	0.888	0.891
		Mean	0.902	0.866
	35	DAL	0.898	0.913
		SSM	0.927	NA
		STM	0.928	NA
		Mean	0.918	NA

Table 6. Local Air Temperature VS. Downstream Water Temperature



Buffer Wdith		Site	2014 Kendall's Tau	2015 Kendall's Tau
	0	ELB	0.55	0.497
		STB	0.584	0.585
		VSB	0.637	0.449
		VVB	0.43	0.745
		Mean	0.55	0.569
	15	SSY	0.63	NA
		STA	0.383	0.535
		VPL	0.54	0.557
		VSA	0.639	0.359
		VVA	0.523	0.765
		DEG	0.729	0.65
		ELA	0.513	0.595
		Mean	0.565	0.577
	35	DAL	0.645	0.558
		SSM	0.651	NA
		STM	0.523	NA
		Mean	0.606	NA

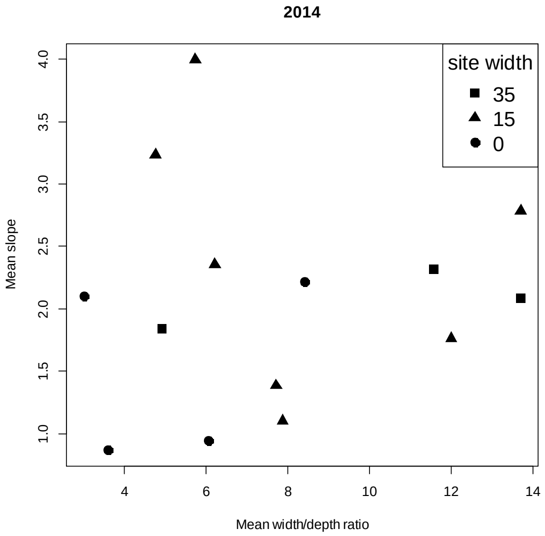


Figure 8. Relationship of the mean weekly slope and width to depth ratio at each site by site width in 2014.

a. Data comes from the third week of the study period (July 11-17)
b. Mean weekly slope, n=7 days
c. Width to depth ratio, n=3 width measurements/ n=3 thalweg measurments at the same points in the stream.

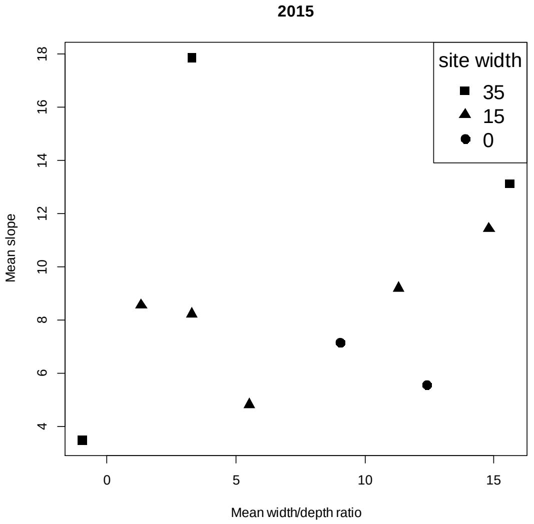


Figure 9. Relationship of the mean weekly slope and width to depth ratio at each site by site width in 2015.

d. Data comes from the second week of the study period (July 4-10)
e. Mean weekly slope, n=7 days
f. Width to depth ratio, n=3 width measurements/ n=3 thalweg measurments at the same points in the stream.
g. Missing data from sites; STM 35’, SSM 35’, VPL 15’, SSY 15’

Table.7 Percent abundance in 2014 and 2015 by species at each site corrected for length of reach sampled.

		VVB	VSB	ELB	STB	VVA	SSY	VPL	VSA	ELA	STA	DGR	SSM	STM	DAL
	year	0'	0'	0'	0'	15'	15'	15'	15'	15'	15'	15'	35'	35'	35'
Chinook salmon	2014	0	0	0	0	0	0	0	0	0	0	0	0	7	0
	2015	0	0	0	0	0	0	0	0	8	0	0	0	0	0
coho salmon	2014	11	0	2	3	27	38	17	0	4	4	55	50	7	0
	2015	0	0	2	15	0	39	0	0	8	0	25	28	18	0
cutthroat trout	2014	11	33	3	24	38	32	50	0	8	22	9	45	21	0
	2015	50	2	4	40	83	43	30	8	38	33	11	65	27	0
rainbow trout	2014	0	0	2	0	0	0	0	0	0	0	0	0	0	0
	2015	0	0	0	0	0	0	0	0	0	17	0	1	9	0
sculpin spp.	2014	0	0	2	0	0	0	17	0	0	0	0	0	0	0
	2015	0	0	1	1	0	0	70	0	0	0	0	0	9	0
largescale sucker	2014	0	0	0	0	0	0	0	0	2	0	0	0	0	0
	2015	0	0	0	0	0	0	0	0	31	0	0	0	0	0
lamprey ammocoetes	2014	0	0	2	1	0	12	0	0	2	0	0	0	0	59
	2015	50	2	0	7	5	7	0	0	0	0	7	6	0	56
western brook lamprey	2014	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2015	0	1	0	0	0	0	0	0	0	0	0	0	9	0
three-spined stickleback	2014	78	67	91	72	27	18	17	100	83	74	36	5	50	41
	2015	0	94	88	35	11	11	0	92	15	50	57	0	27	44

Table 7 continued.

		VVB	VSB	ELB	STB	VVA	SSY	VPL	VSA	ELA	STA	DGR	SSM	STM	DAL
	year	0'	0'	0'	0'	15'	15'	15'	15'	15'	15'	15'	35'	35'	35'
bluegill	2014	0	0	0	0	0	0	0	0	0	0	0	0	7	0
	2015	0	0	1	0	2	0	0	0	0	0	0	0	0	0
pumpkinseed	2014	0	0	0	0	8	0	0	0	2	0	0	0	7	0
	2015	0	0	3	1	0	0	0	0	0	0	0	0	0	0
brown bullhead	2014	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2015	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Total number of fish (corrected by length of reach=90)	2014	10	8	64	180	36	50	6	12	54	23	101	38	14	22
	2015	6	86	129	84	66	61	10	38	13	7	28	68	11	42

a. Length of reach sampled was 90 meters except in 2015 when; VVB=60 m, ELB=84, DAL=87.

b. Proper common names of fish are used (Robins et al. 1991)

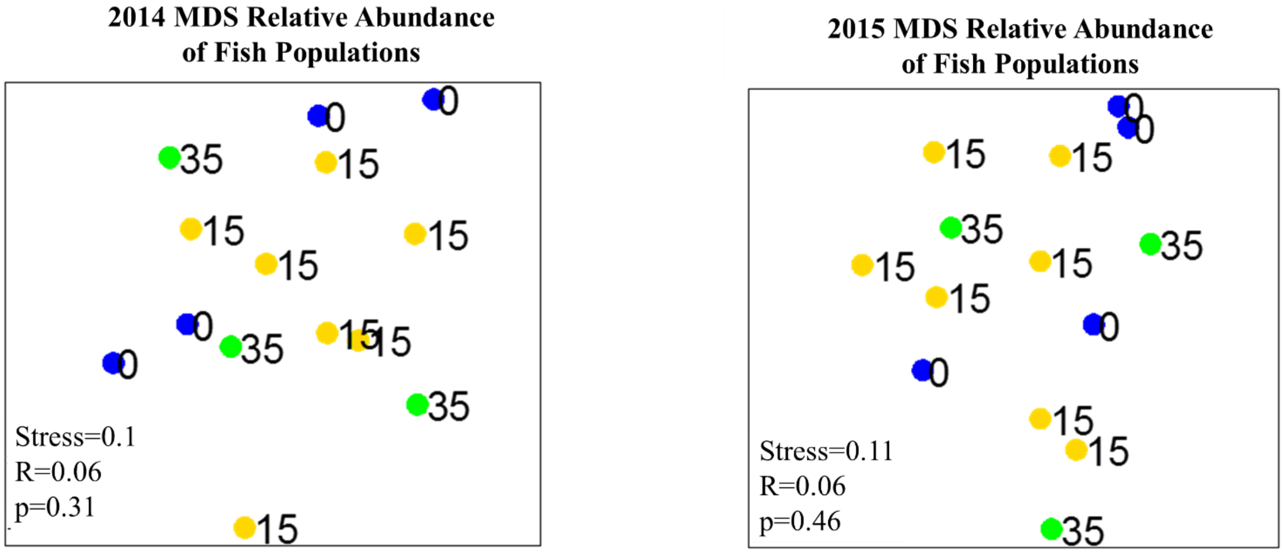
Table 8. Mean % relative abundance for the most numerous three species of fish among site widths in 2014 and 2015.

Fish Assemblage Mean Percent Relative Abundance					
Site Width	Fish Species	2014		2015	
	three-spined stickleback	70.6[a]	16.6[b]	54.3	45.1
0'	coho salmon	3.7	4.4	4.3	7.5
(N=4)	cutthroat trout	9.3	10.6	24.2	24.6
	sum	83.5[c]		82.8	
	three-spined stickleback	50.5	33.9	32.8	33
15'	coho salmon	20.8	20.6	10.3	15.8
(N=7)	cutthroat trout	21.3	17.4	34.3	24.7
	sum	92.7		77.4	
	three-spined stickleback	32.1	23.6	23.7	22.2
35'	coho salmon	19	27	15.4	14.2
(N=3)	cutthroat trout	22.1	22.4	30.7	32.5
	sum	73.1		69.8	

a Percent relative abundance = $\frac{\#\text{fish species of one type at site A}}{\text{total \# fish of all species at site A}} * 100$

b. Standard deviation.

c. Sums of the relative abundance of the three species combined for each buffer width is proportion of the entire population that they represent.



2014 MDS Relative Abundance of Fish Populations

2015 MDS Relative Abundance of Fish Populations

Figure 10: Multi-dimensional scaling representation for 2014 (left) and 2015 (right) relative abundance of fish assemblages.

a. Each point on the figure represents a buffer site.

b. Blue dots=0 buffer and represent sites with no buffer, yellow dots =15-foot, and green dots =35-foot widths

Table 9: Vegetation characteristics of sites by buffer width.

	Vegetation Characteristics						
Sites per Buffer Width	Mean # Species per buffer (stdev)	Mean plant density #stems/m^2 (stdev)	Mean DBH inches (stdev)	Mean height meters (stdev)	2014 Mean % Effective Shade (stdev)	2015 Mean % Effective Shade (stdev)	Dominant species
0' (n=4)	2	NA	NA	NA	8.5 (7.39) n=71	14.4 (10.7) n=62	reed canary grass and Himalayan blackberry
15' (n=6)	12.5 (6.73)	0.62 (0.36)	5.15 (6.73)	3.37 (2.83)	74.8 (27.3) n= 209, 7sites	83.0 (26.7) n=210	Rosa spp. (5/6 sites)
35' (n=3)	14 (2.65)	0.22 (0.1)	6.57 (6.58)	3.85 (2.78)	83.4 (40.9) n=69	85 (26.0) n=90	Willow (1/3), Red osier dogwood (1/3), Douglas fir (1/3)

a. The means of the measurements; number of species, plant density, diameter at breast height, height, and effective shade, are given, followed by standard deviation in parentheses.

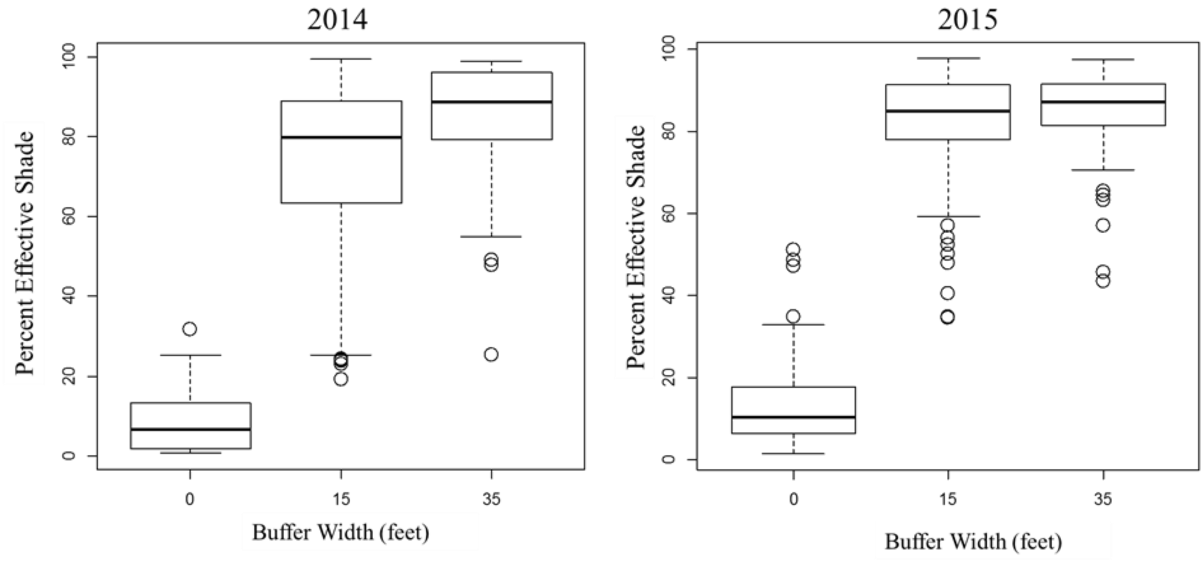


Figure 11. Boxplots showing percent effective shade between site widths in 2014 (left) and 2015 (right).

a. The thick line in the center of the boxes represent the median shade measurement.
b. Circles are outliers.

Table 10. Stream characteristic summary by site width, in 2014 and 2015.

a. Mean stream flow rate, thalweg, width, and dominant substrate type are listed for sites by width

b. Standard deviation

c. Sample size of each measurement is variable and is located within the table in its own column.

Stream Characteristics

Year	Sites by Buffer Width	Mean Flow Rate m^3/s		Mean Thalweg meters		Mean Width meters		Dominant Substrate n=50/site
2014	0'	0.07[a] N=12[c]	0.05[b]	1.6 N=14	0.43	2.72 N=14	1.4	Fine Sediment 3/4 of sites
	15'	0.1 N=21	0.09	1.4 N=28	0.67	2.54 N=28	1.0	Fine Sediment 6/7 sites
	35'	0.04 N=9	0.02	1.2 N=14	0.9	1.87 N=14	0.74	Sand 1/3, Coarse gravel 1/3, Fine Sediment 1/3 of sites
2015	0'	0.05 N=12	0.05	0.4 N=12	0.17	2.27 N=12	0.92	Fine Sediment 3/4 sites
	15'	0.05 N=21	0.05	0.3 N=21	0.2	2.38 N=21	0.84	FineSediment 5/7 sites
	35'	0.02 N=9	0.01	0.3 N=9	0.22	2.01 N=9	0.53	Sand 3/3 sites

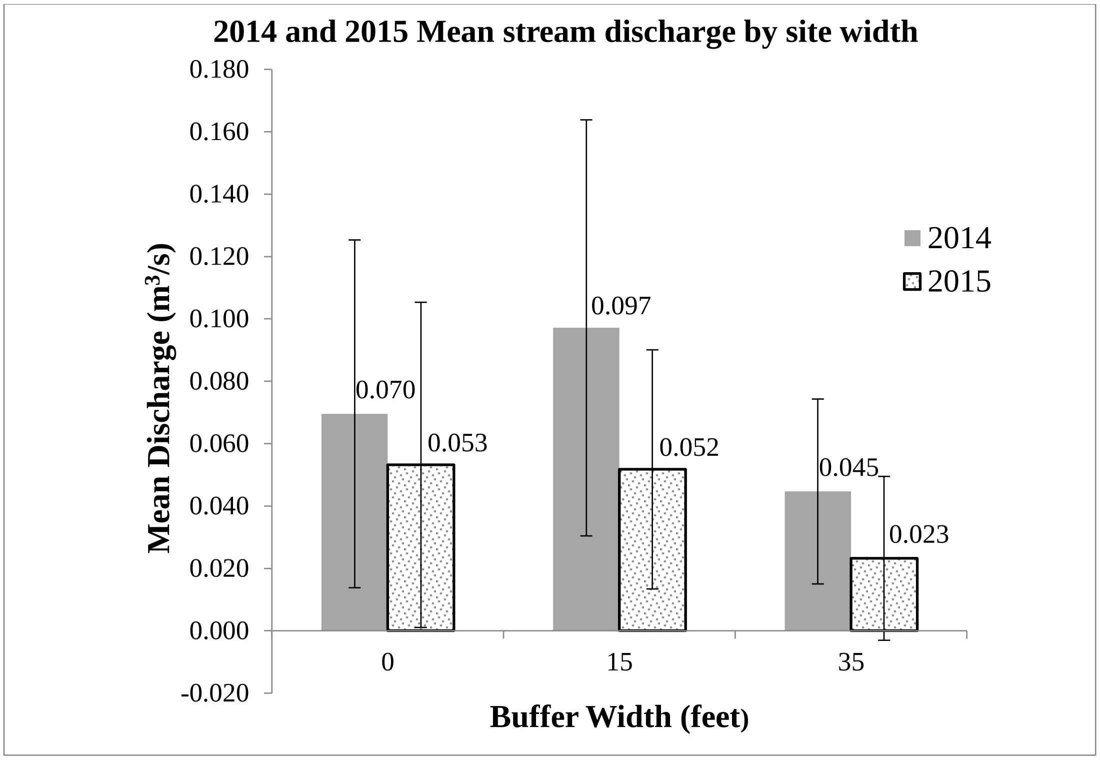


Figure 12. Mean stream flow (m^3/s) by buffer width in 2014 (left) and 2015 (right).

a. Bars show 95% confidence intervals.

b. The mean of three measurements were taken at each site; n=4 for 0'buffer sites, n=7 for 15' buffer sites, and n=3 for 35' buffer sites.

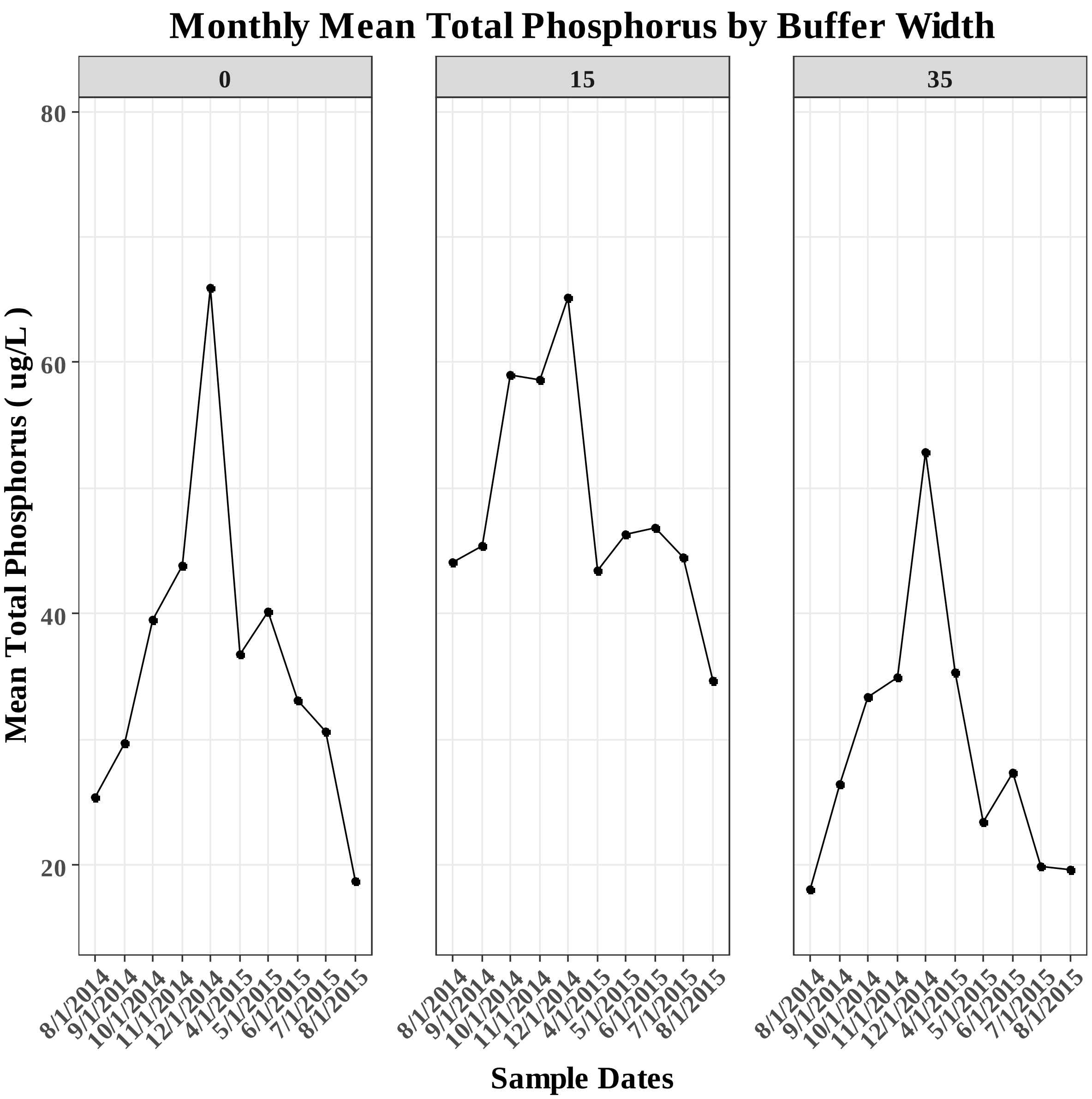


Figure 13. Mean total phosphorus measurements in parts per billion (ug/L) grouped by buffer width category at each site.

a. Shaded grey area represents 95% CI.

b. Sample size: 0’ n=4, except for 8/1/2014 and 4/1/2015 where n=3; 15’n=7, except for 4/1/2015 where n=5; and 35’n=3.

c. Note a gap in sampling occurs at all sites January 2015 –March 2015

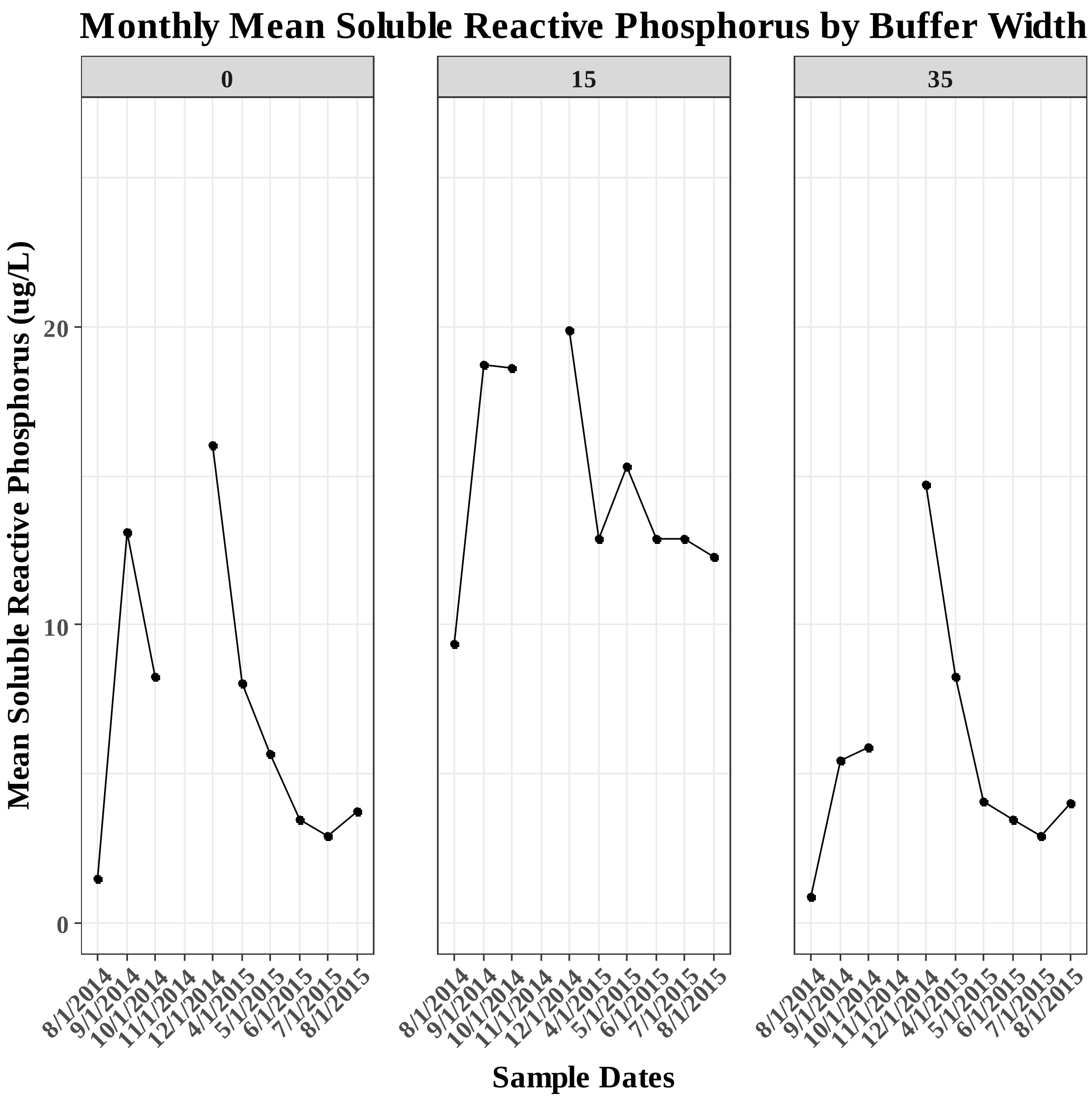


Figure 14. Mean soluble reactive phosphorus measurements in parts per billion (ug/L) grouped by buffer width category at each site.

a. Shaded grey area represents 95% CI.

b. Sample size: 0’ n=4, except for 4/1/2015 where n=3; 15’n=7, except for 4/1/2015 where n=6; and 35’ n=3.

c. Note a gap in sampling occurs at all sites January 2015 –March 2015

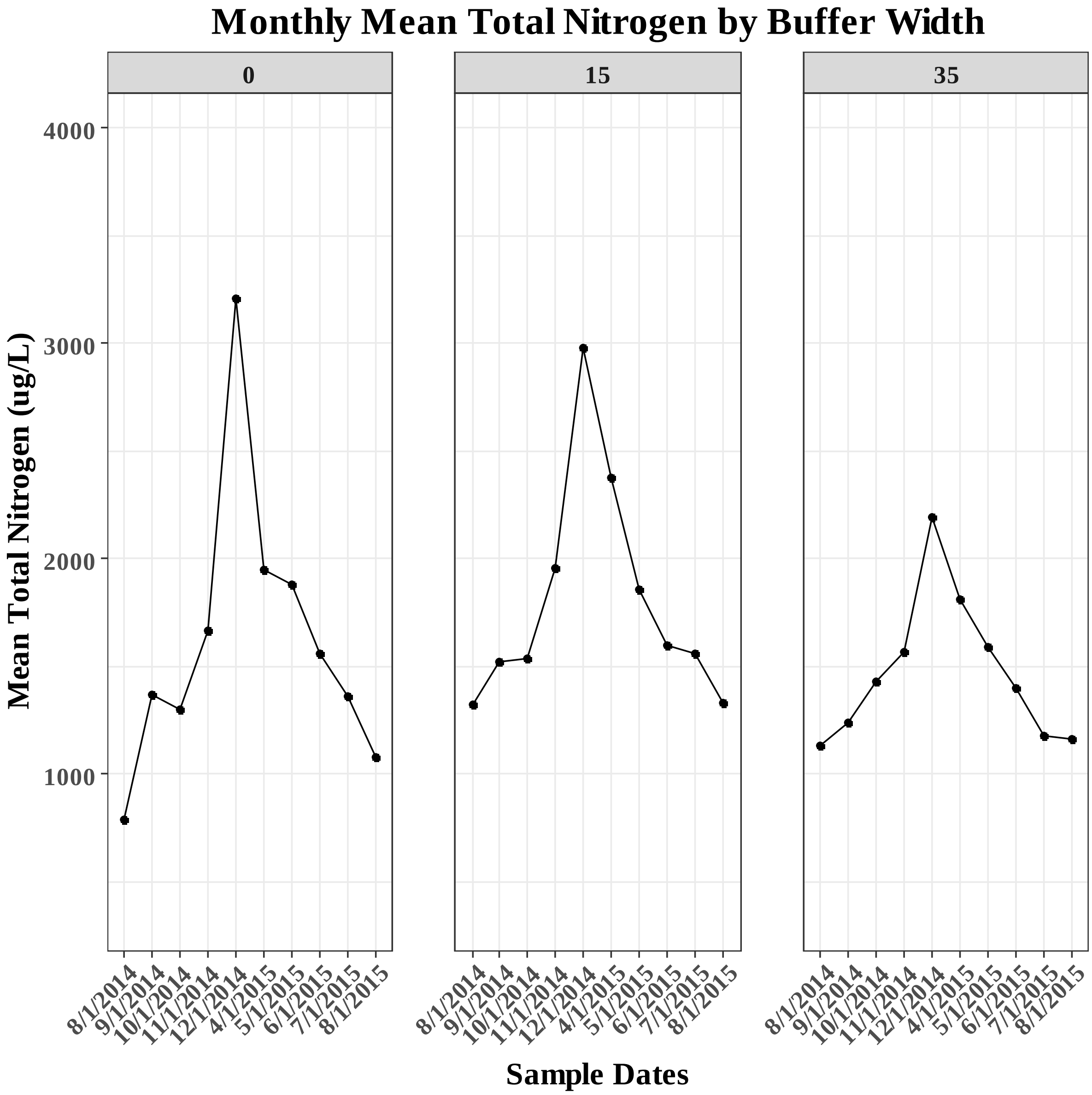


Figure 15. Mean total nitrogen measurements in parts per billion (ug/L) grouped by buffer width category at each site.

a. Shaded grey area represents 95% CI.

b. Sample size: 0’ n=4, except for 8/1/2014 and 4/1/2015 where n=3; 15’ n=7, except for 4/1/2015 where n=5; and 35’ n=3.

c. Note a gap in sampling occurs at all sites January 2015 –March 2015

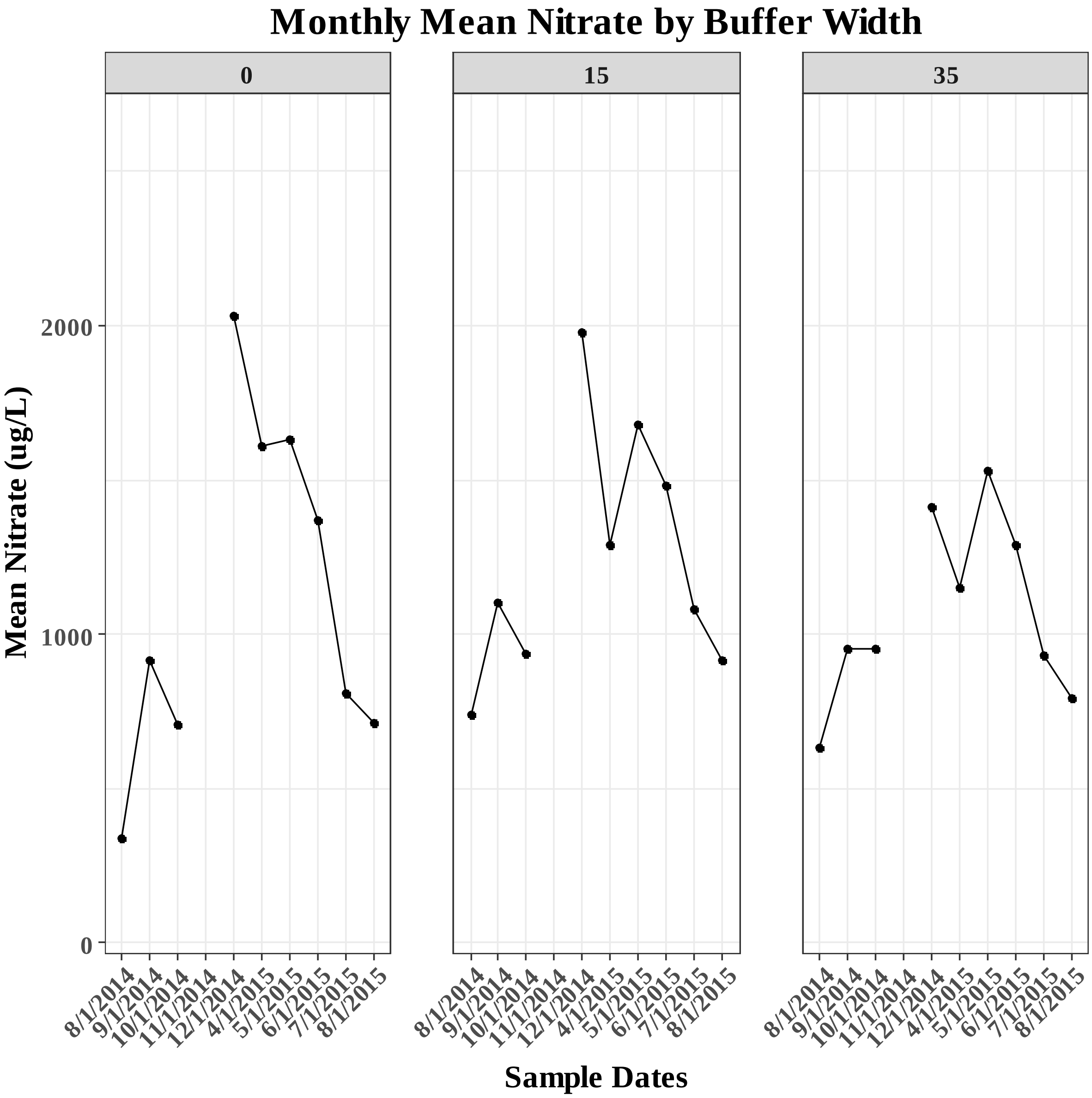


Figure 16. Mean total nitrogen measurements in parts per billion (ug/L) grouped by buffer width category at each site.

a. Shaded grey area represents 95% CI.

b. Sample size: 0’ n=4, except for 8/1/2014 and 4/1/2015 where n=3; 15’ n=7, except for 4/1/2015 where n=6; and 35’ n=3.

c. Note a gap in sampling occurs at all sites January 2015 –March 2015

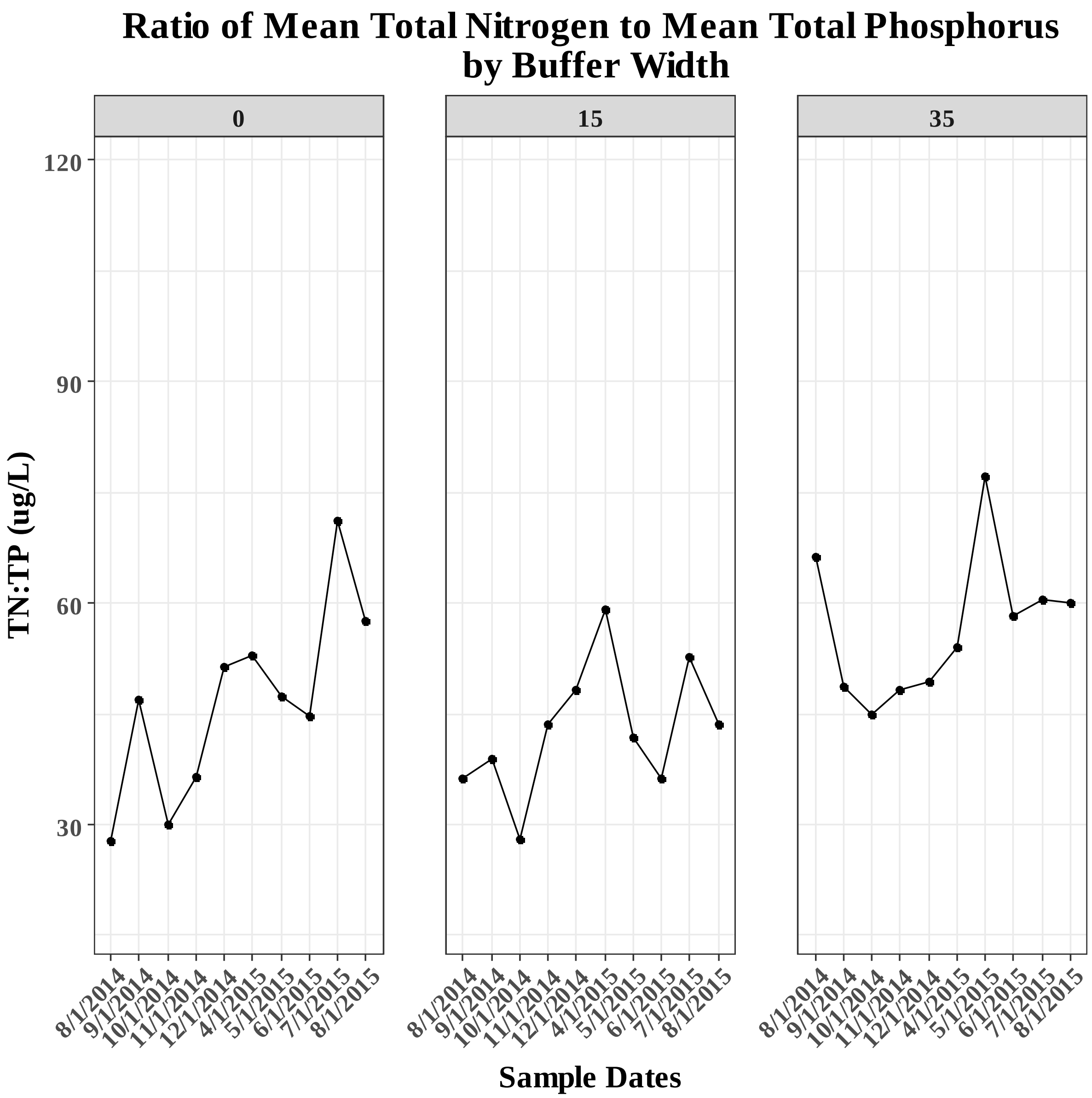


Figure 17. Ratio of total nitrogen to total phosphorus measurements in parts per billion (ug/L) grouped by buffer width category.

a. Shaded grey area represents 95% CI.

b. Sample size: 0 buffer n=4, except for 8/1/2014 and 4/1/2015 where n=3; 15-foot buffer n=7, except for 4/1/2015 where n=5; and 35-foot buffer n=3.

c. Note a gap in sampling occurs at all sites January 2015 –March 2015

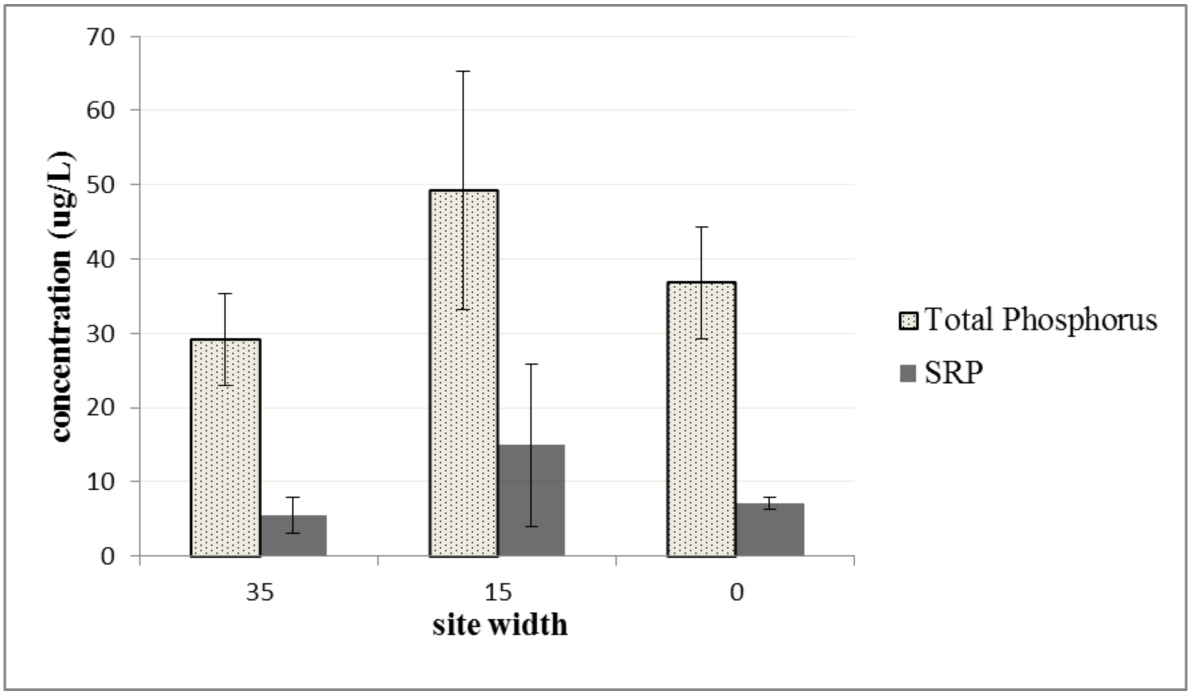


Figure18 Mean annual concentration of total phosphorus and soluble reactive phosphorus.

a. Data are monthly samples August-December 2014 and April-August 2015
b. Bars are the 95% confidence intervals

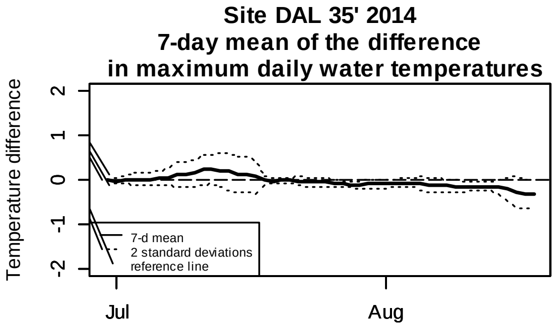
Figure 19. Site VVA, a 15’ buffer, with a pool under a small beaver dam.

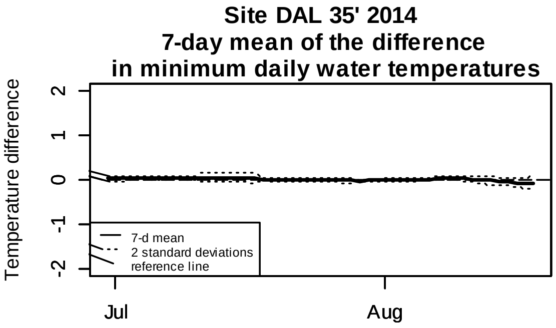
Appendix I. The 7-day rolling mean of the differences, from upstream to downstream, in daily maximum (left column) and daily minimum (right column) water temperatures.

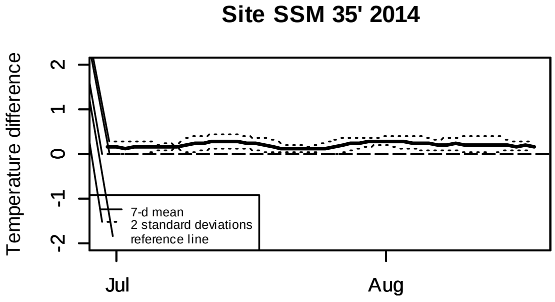
Note: Positive differences indicate stream warming, negative differences indicate stream cooling.

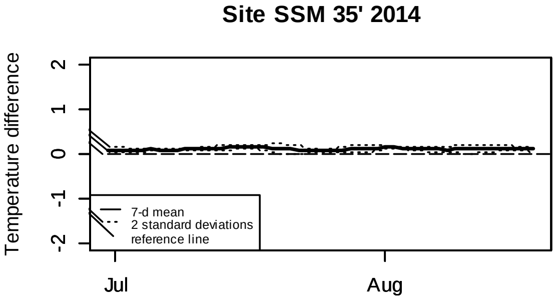
Rolling means of the temperature differences are shown for the study period June 27-August 21 in 2014 and 2015.

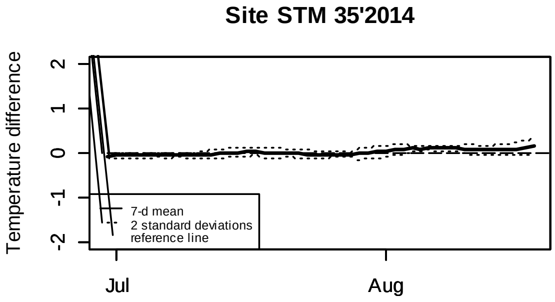
Figures are given for each individual site and are grouped chronologically by year.

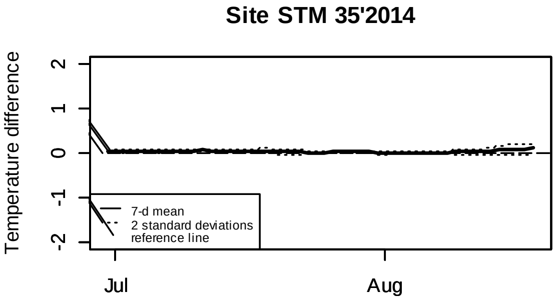


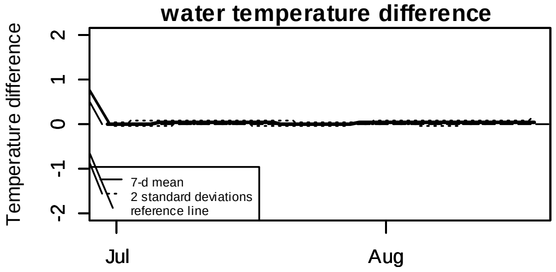




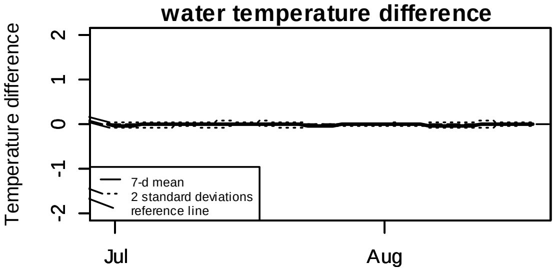




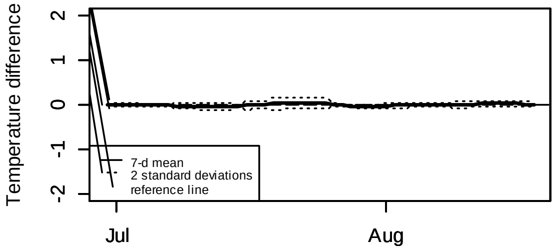




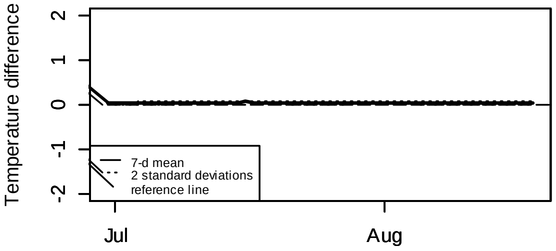
Site DGR 15' 2014
7-day mean of the daily maximum
water temperature difference



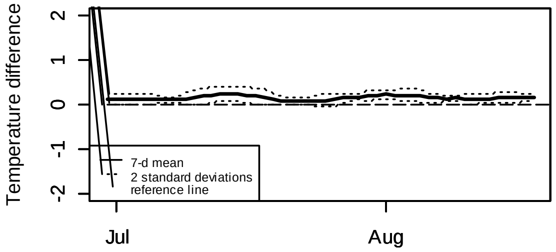
Site DGR 15' 2014
7-day mean of the daily minimum
water temperature difference



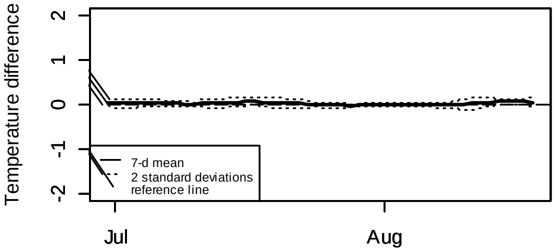
Site ELA 15' 2014



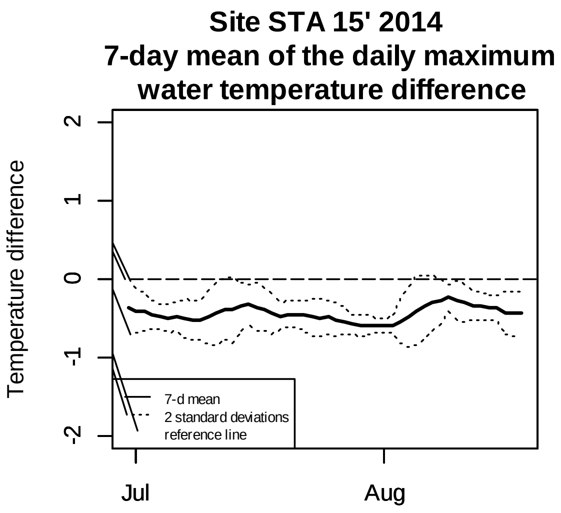
Site ELA 15' 2014

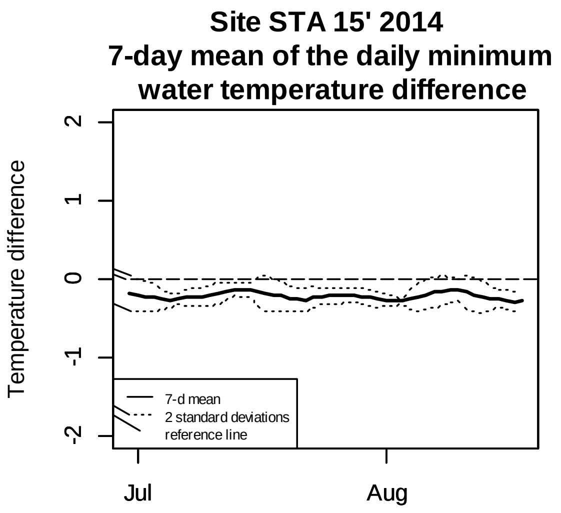


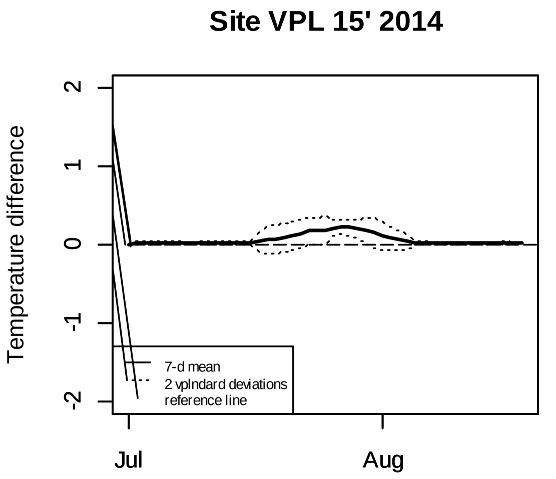
Site SSY 15' 2014

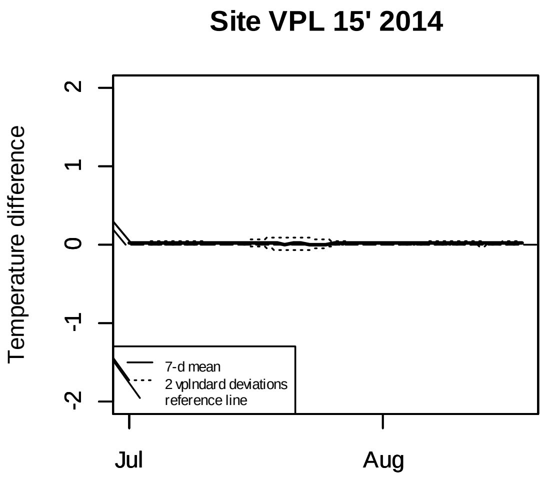


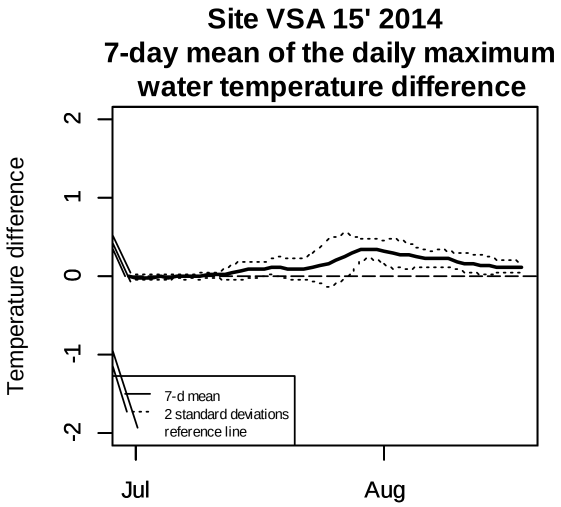
Site SSY 15' 2014

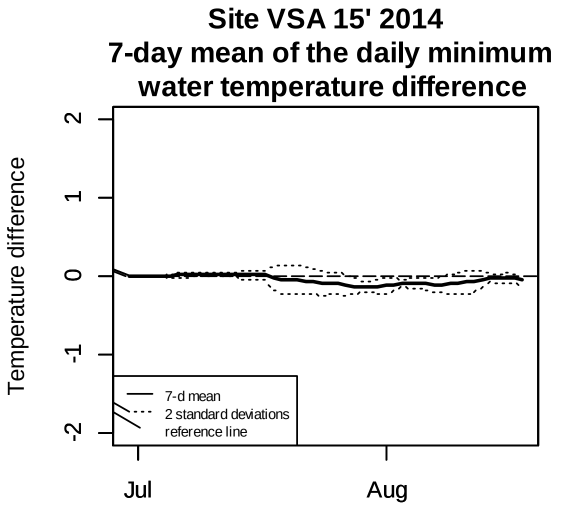

Site STA 15' 2014
7-day mean of the daily maximum
water temperature difference

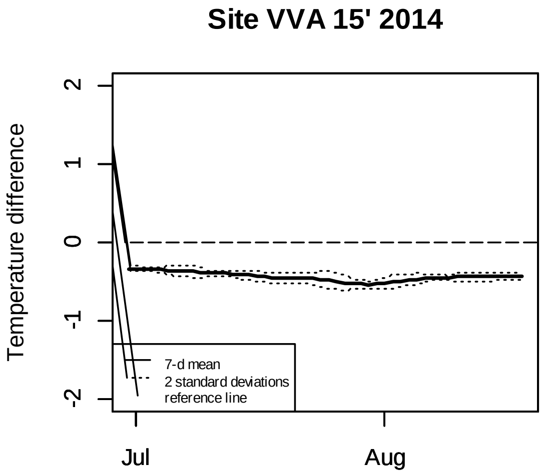

Site STA 15' 2014
7-day mean of the daily minimum
water temperature difference

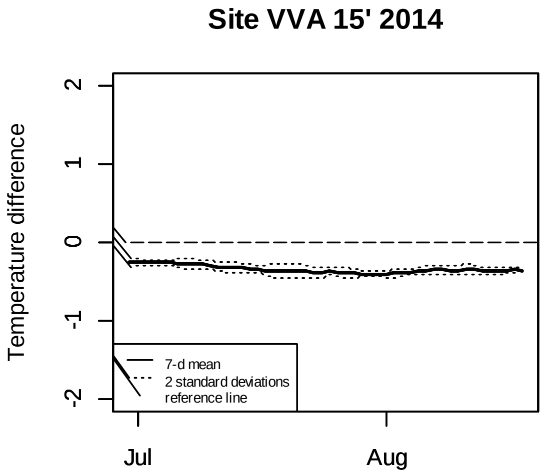

Site VPL 15' 2014

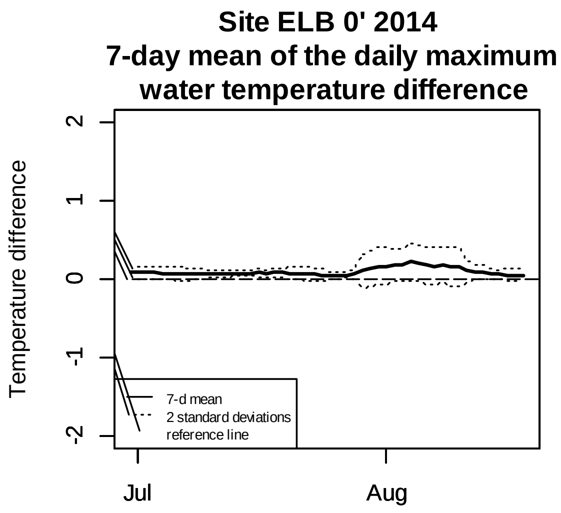

Site VPL 15' 2014

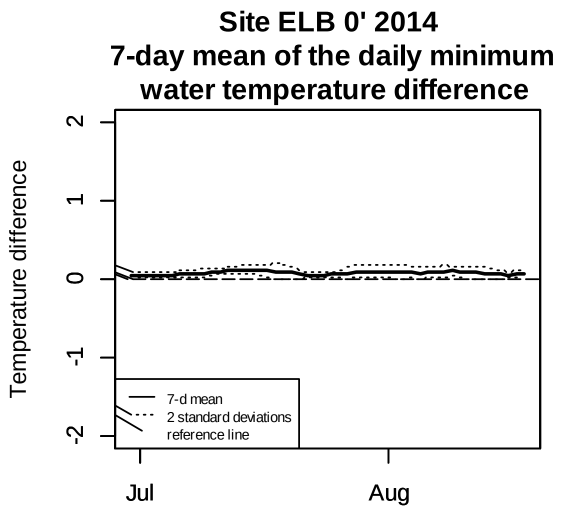

Site VSA 15' 2014
7-day mean of the daily maximum water temperature difference
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

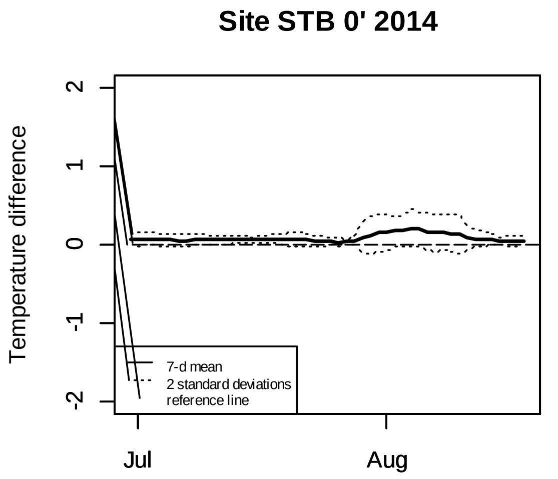

Site VSA 15' 2014
7-day mean of the daily minimum water temperature difference
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

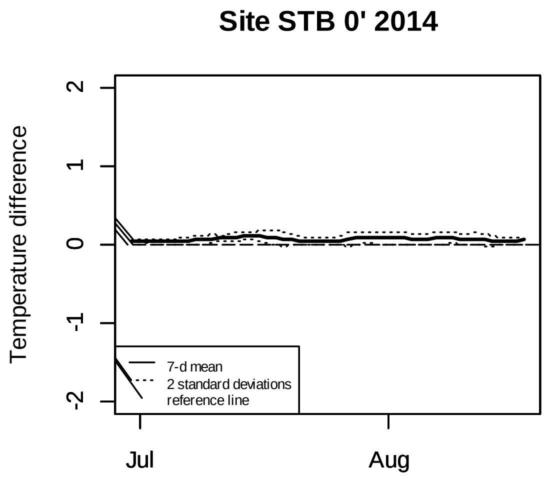

Site VVA 15' 2014
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

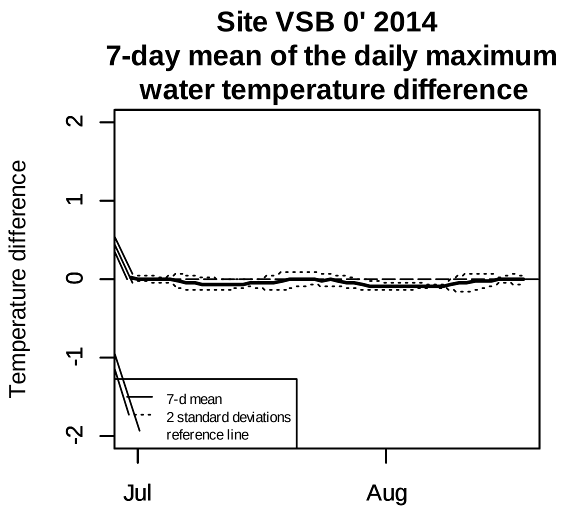

Site VVA 15' 2014
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

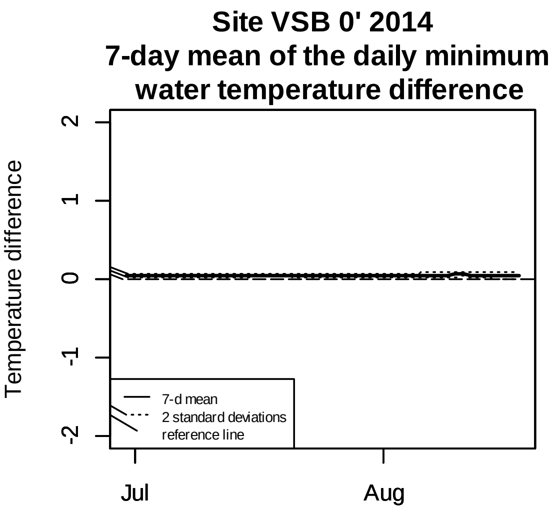

Site ELB 0' 2014
7-day mean of the daily maximum
water temperature difference

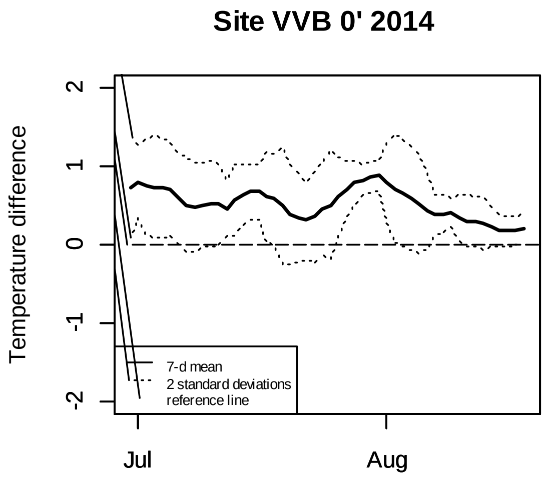

Site ELB 0' 2014
7-day mean of the daily minimum
water temperature difference

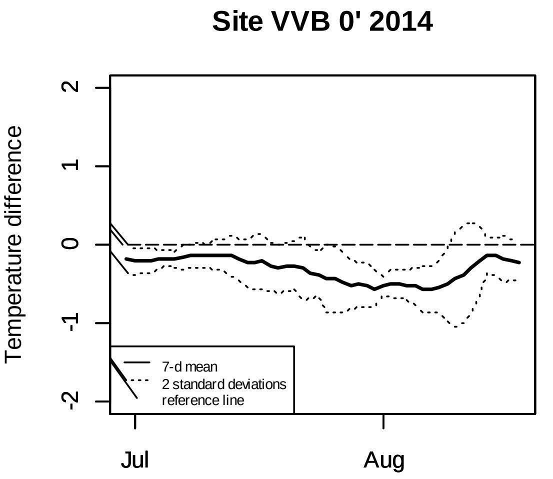

Site STB 0' 2014

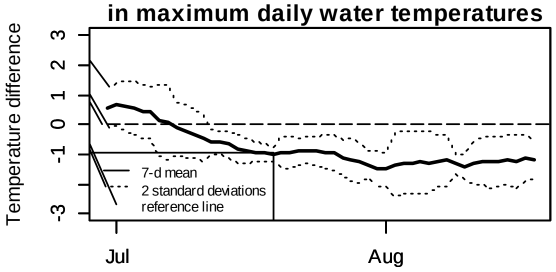

Site STB 0' 2014

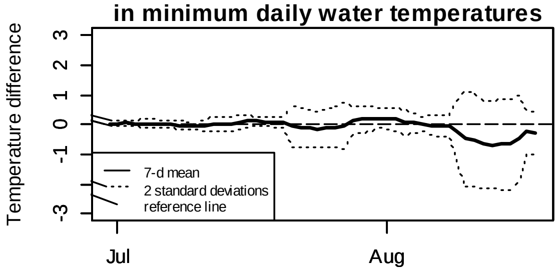

Site VSB 0' 2014
7-day mean of the daily maximum
water temperature difference
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

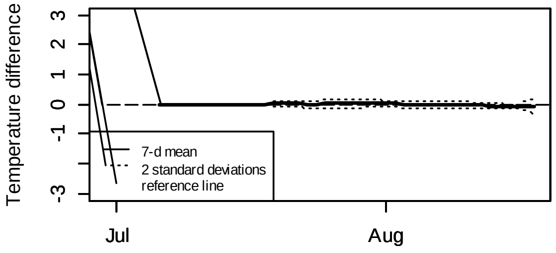

Site VSB 0' 2014
7-day mean of the daily minimum
water temperature difference
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

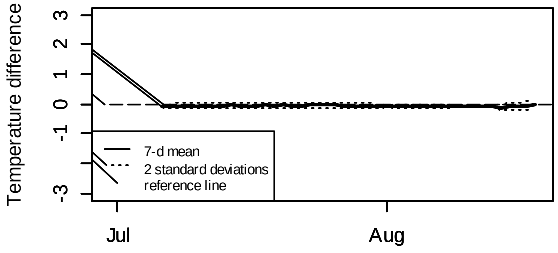

Site VVB 0' 2014
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

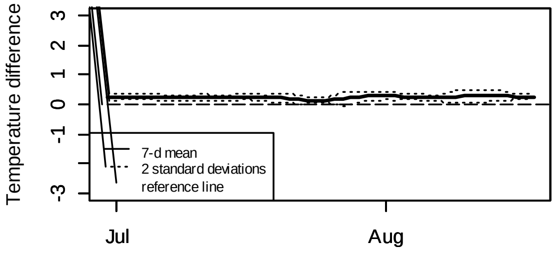

Site VVB 0' 2014
Temperature difference
2
1
0
-1
-2
Jul
Aug
7-d mean
2 standard deviations
reference line

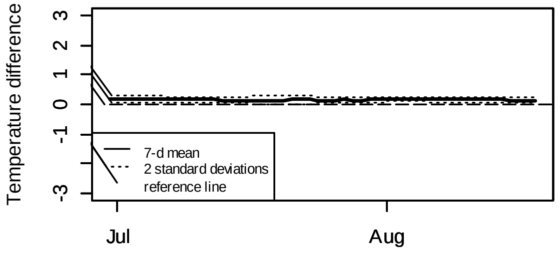

Site DAL 35' 2015
7-day mean of the difference
in maximum daily water temperatures


Site DAL 35' 2015
7-day mean of the difference
in minimum daily water temperatures


Site STM 35'2015


Site STM 35'2015


Site SSM 35' 2015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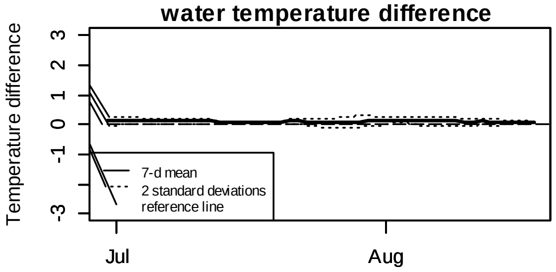
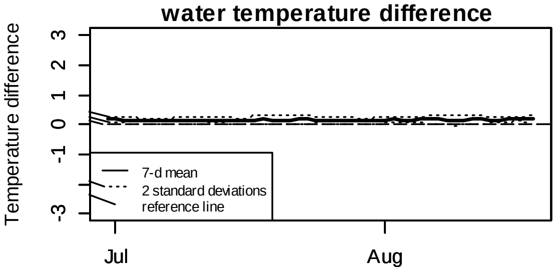
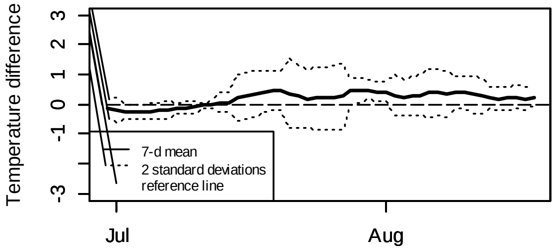
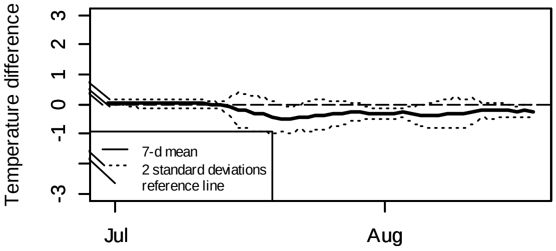
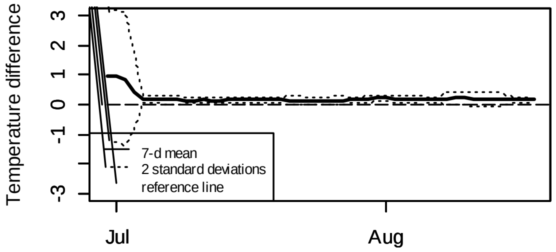
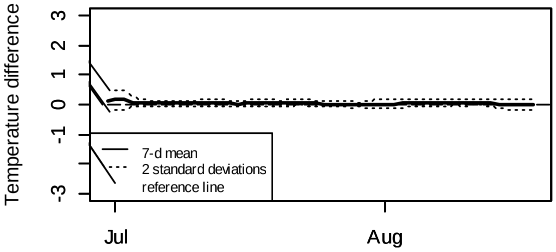

Site DGR 15' 2015
7-day mean of the daily maximum
water temperature difference

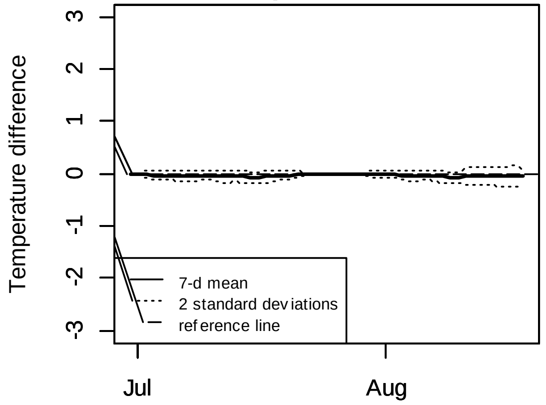

Site DGR 15' 2015
7-day mean of the daily minimum
water temperature difference


Site ELA 15' 2015

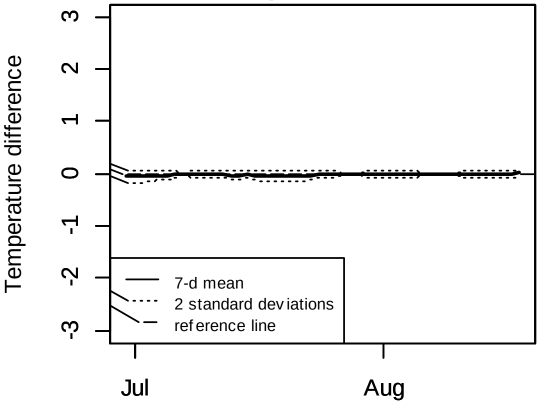

Site ELA 15' 2015


Site SSY 15' 2015

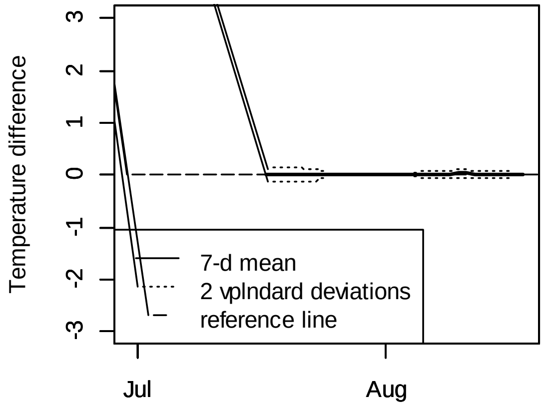

Site SSY 15' 2015



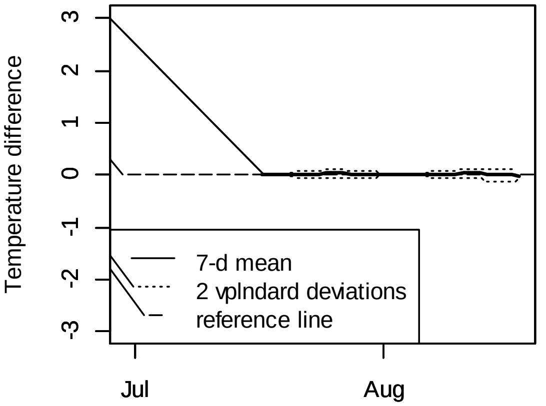
Site STA 15' 2015
7-day mean of the daily maximum
water temperature difference



Site STA 15' 2015
7-day mean of the daily minimum
water temperature difference

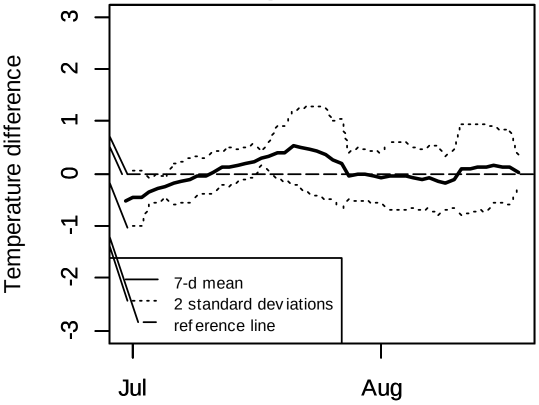


Site VPL 15' 2015

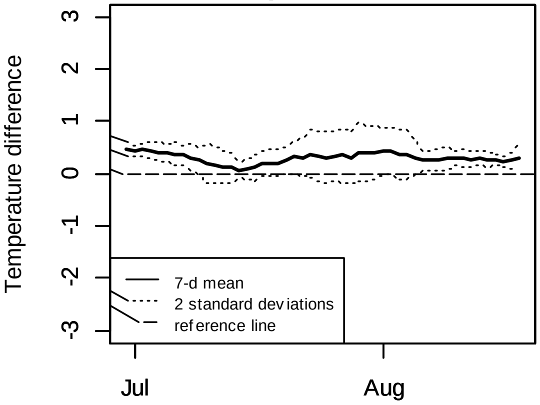


Site VPL 15' 2015

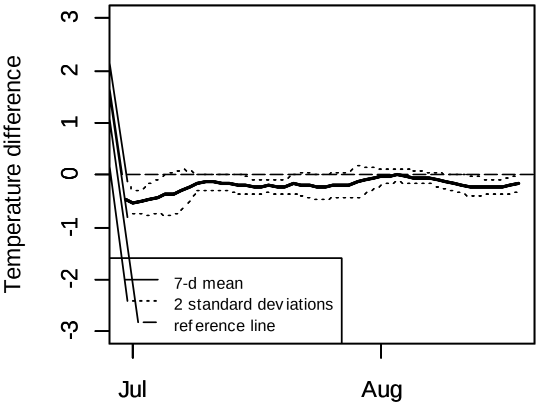
Site VSA 15' 2015
7-day mean of the daily maximum water temperature difference



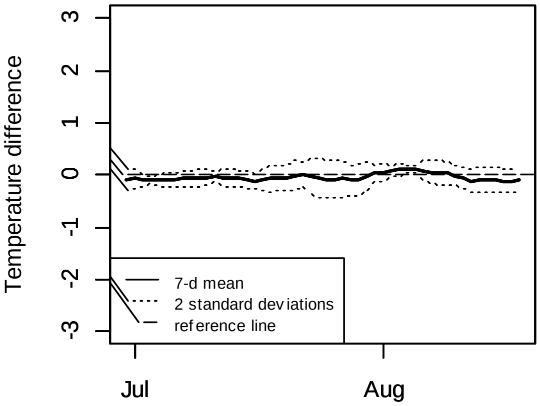
Site VSA 15' 2015
7-day mean of the daily minimum water temperature difference



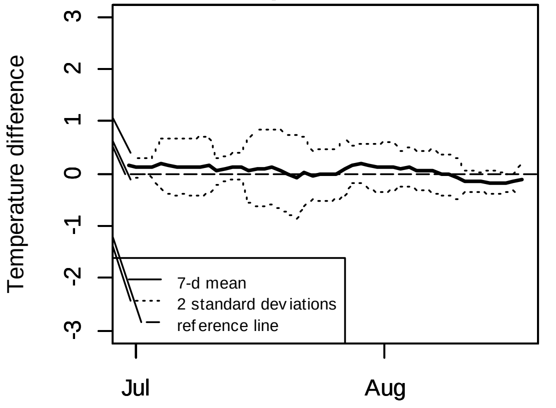
Site VVA 15' 2015



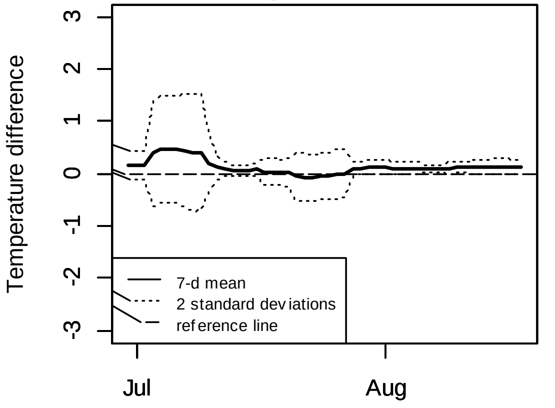
Site VVA 15' 2015



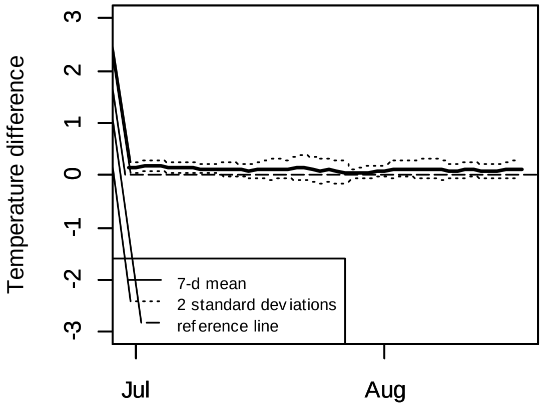
Site ELB 0' 2015
7-day mean of the daily maximum
water temperature difference



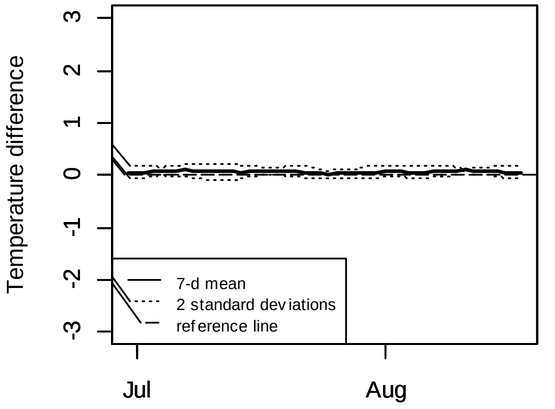
Site ELB 0' 2015
7-day mean of the daily minimum
water temperature difference

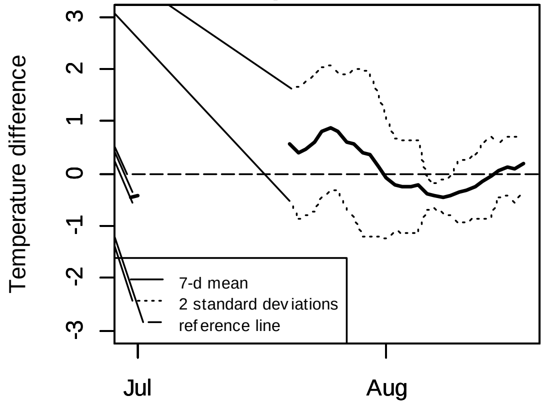


Site STB 0' 2015

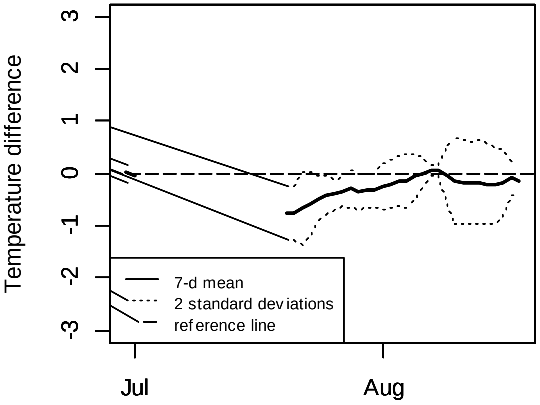


Site STB 0' 2015

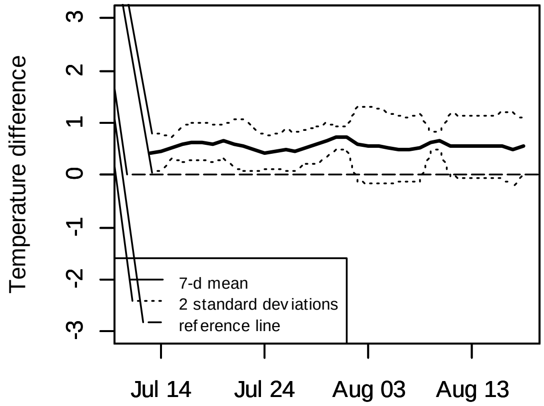




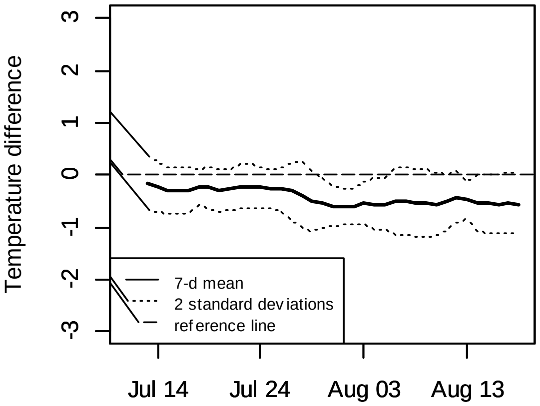
Site VSB 0' 2015
7-day mean of the daily maximum water temperature difference



Site VSB 0' 2015
7-day mean of the daily minimum water temperature difference



Site VVB 0' 2015



Site VVB 0' 2015

Appendix II. Study site photos.

Figure 1. Example of a 15’, densely planted riparian buffer at site VVA. Vegetation; *Rosa spp*., ninebark, red osier dogwood, *Salix spp*.

Figure 2. Example of a 35’ riparian buffer at site DAL. Vegetation; *Salix spp*., red osier dogwood, ninebark.

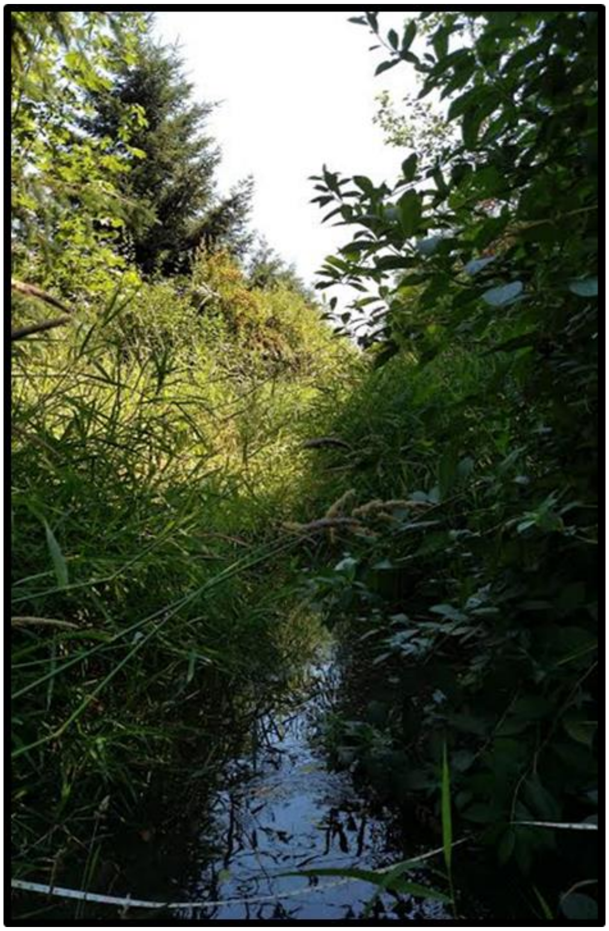
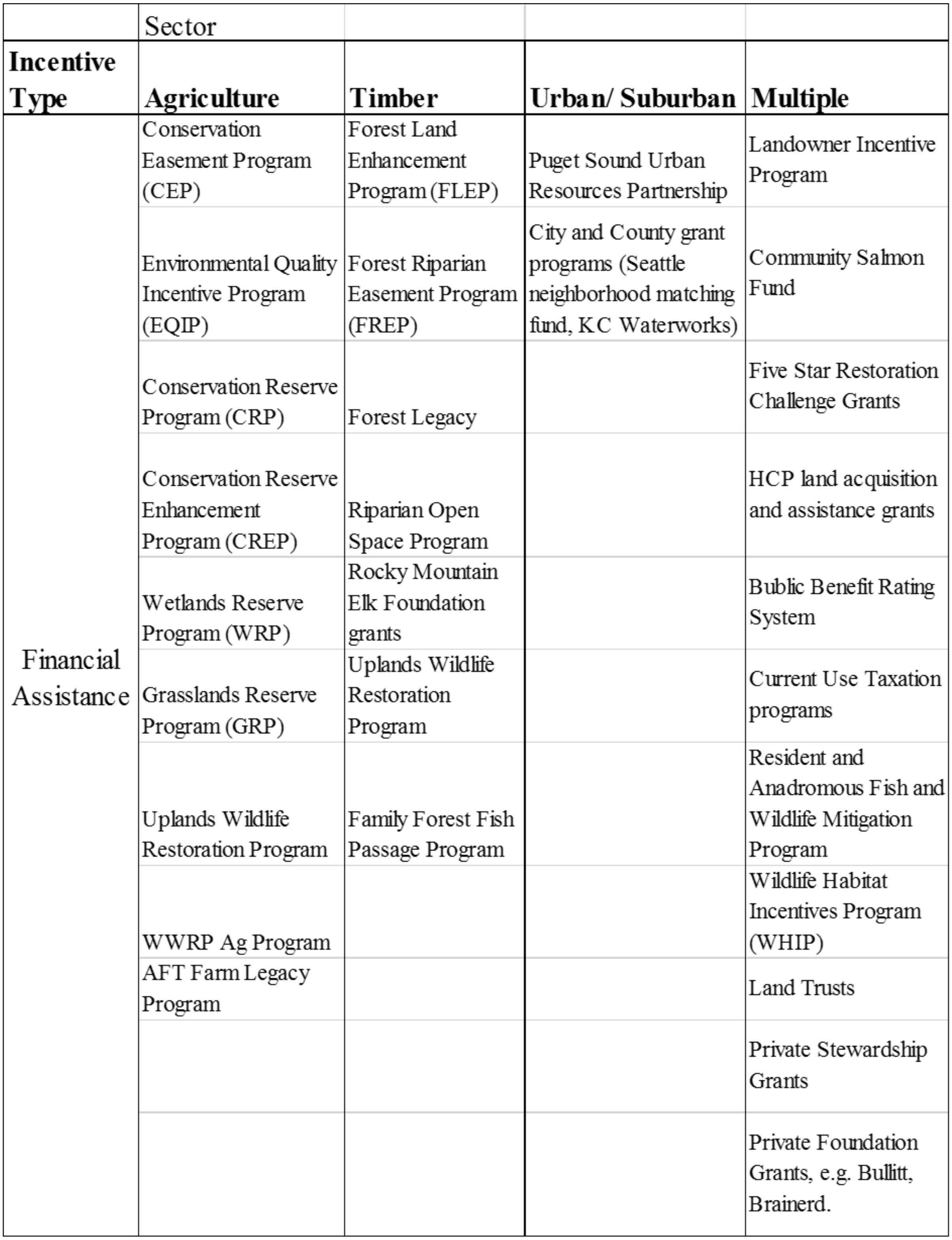


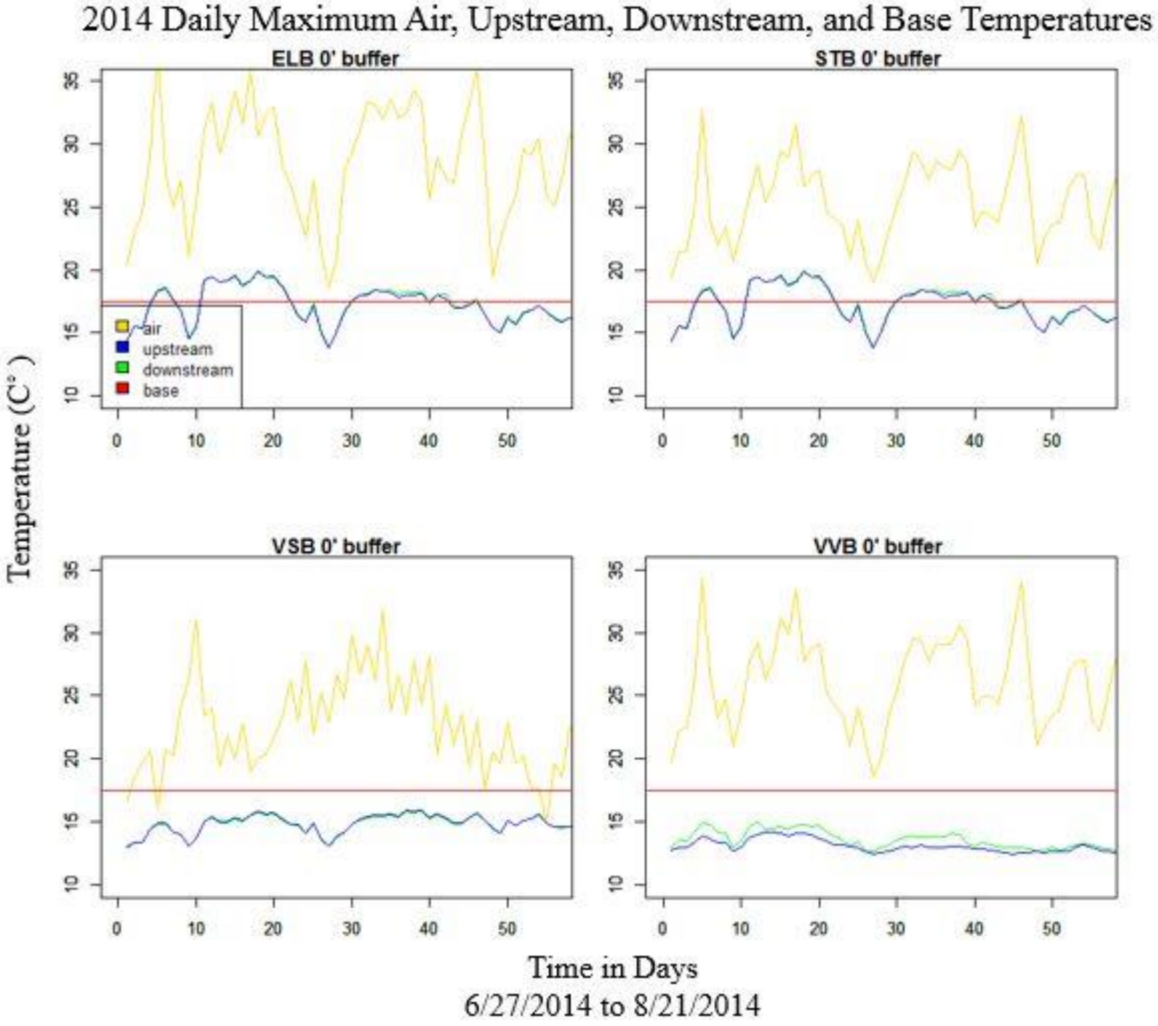
Figure 3. Example of a 35’ riparian buffer at site STM. Vegetation; black twinberry, big leaf maple, Douglas fir, *Salix spp.*, *Rosa spp*., Sitka spruce, Pacific ninebark.

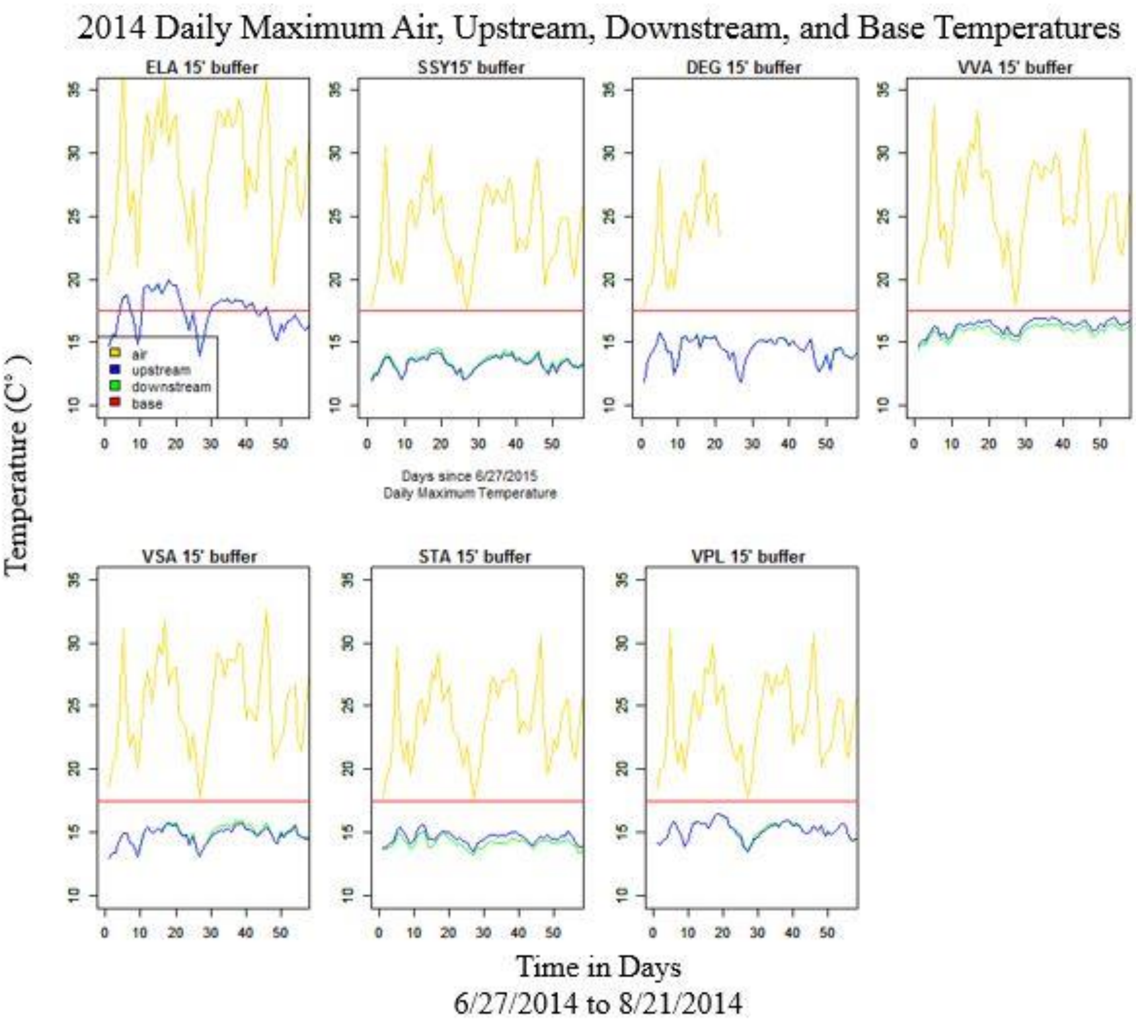
APPENDIX III. Summary of conservation incentive programs by type and sector in Washington State. (from Evergreen Funding Consultants 2007) Note: As of 2016 most programs were consolidated under EQIP

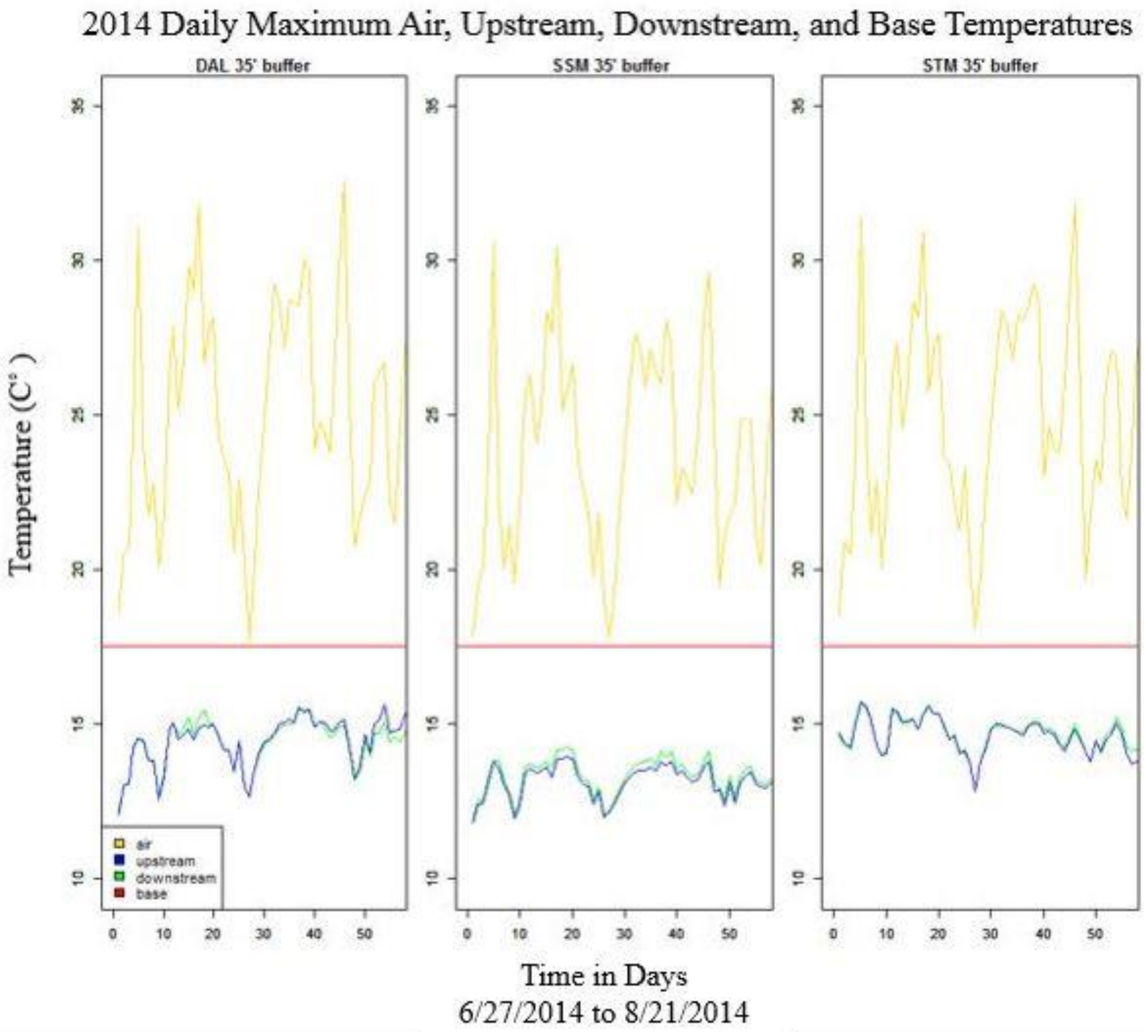


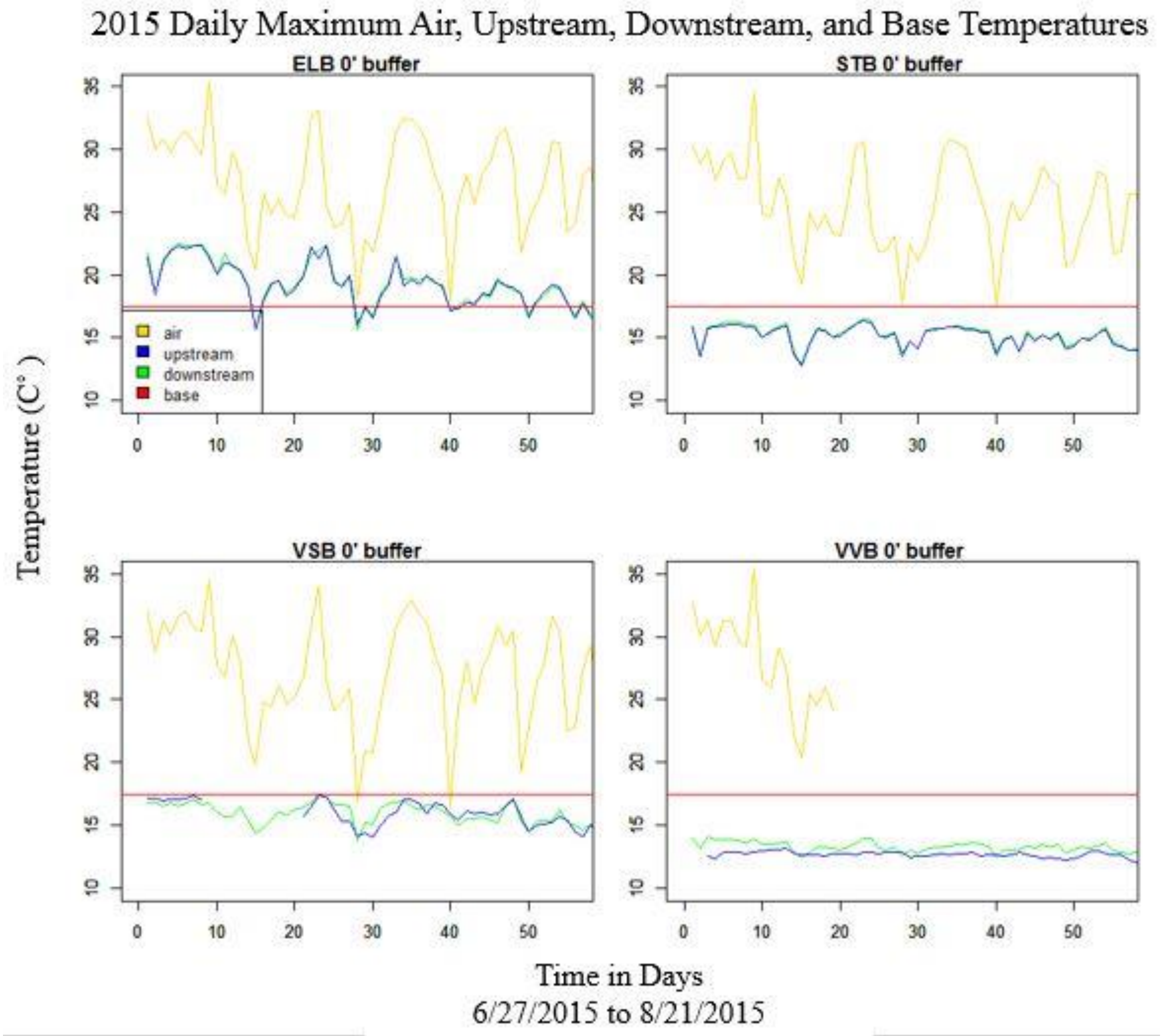
	Sector			
Incentive Type	**Agriculture**	**Timber**	**Urban/ Suburban**	**Multiple**
Financial Assistance	Conservation Easement Program (CEP)	Forest Land Enhancement Program (FLEP)	Puget Sound Urban Resources Partnership	Landowner Incentive Program
	Environmental Quality Incentive Program (EQIP)	Forest Riparian Easement Program (FREP)	City and County grant programs (Seattle neighborhood matching fund, KC Waterworks)	Community Salmon Fund
	Conservation Reserve Program (CRP)	Forest Legacy		Five Star Restoration Challenge Grants
	Conservation Reserve Enhancement Program (CREP)	Riparian Open Space Program		HCP land acquisition and assistance grants
	Wetlands Reserve Program (WRP)	Rocky Mountain Elk Foundation grants		Bublic Benefit Rating System
	Grasslands Reserve Program (GRP)	Uplands Wildlife Restoration Program		Current Use Taxation programs
	Uplands Wildlife Restoration Program	Family Forest Fish Passage Program		Resident and Anadromous Fish and Wildlife Mitigation Program
	WWRP Ag Program			Wildlife Habitat Incentives Program (WHIP)
	AFT Farm Legacy Program			Land Trusts
				Private Stewardship Grants
				Private Foundation Grants, e.g. Bullitt, Brainerd.

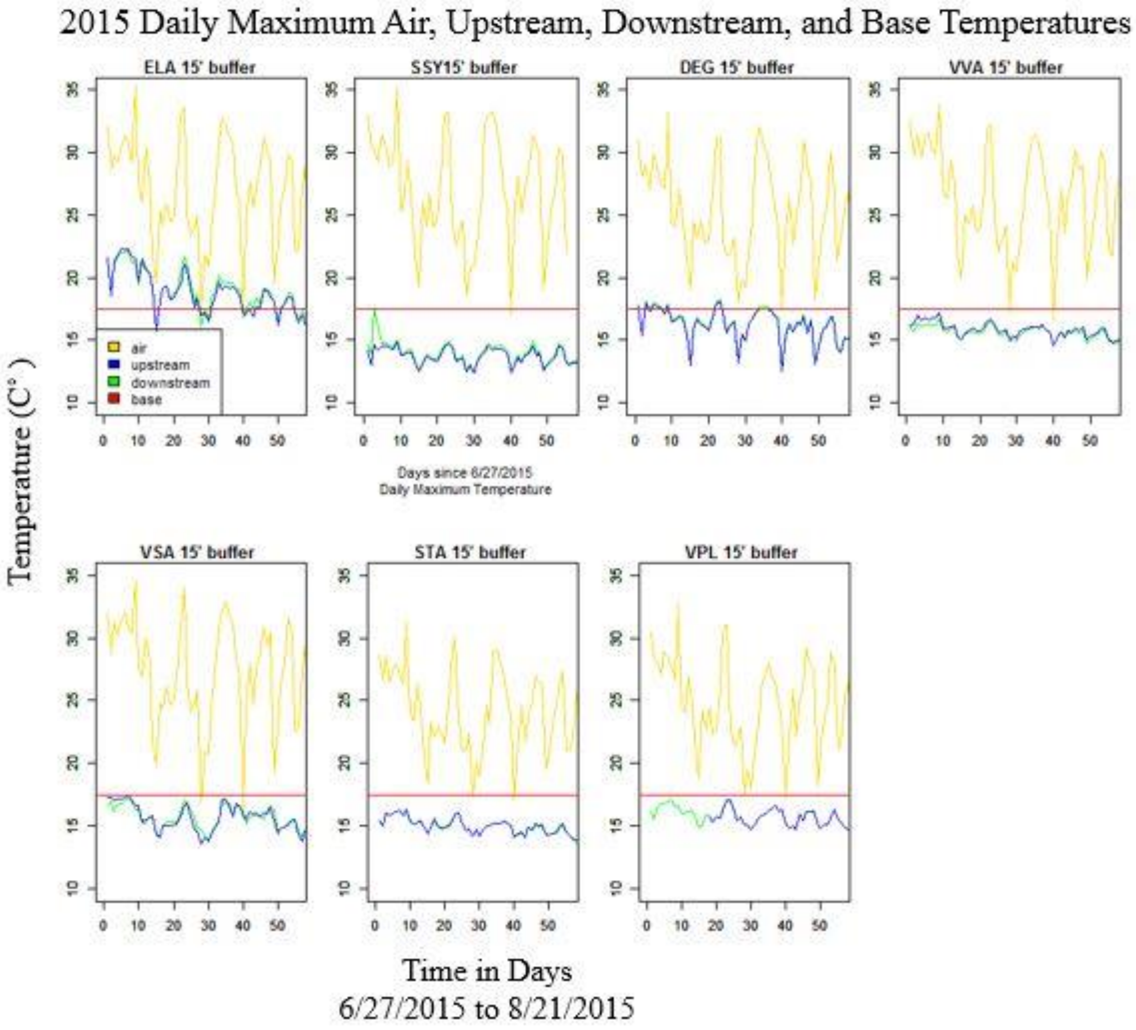
Appendix IV. Daily Maximum Air and Water Temperatures.

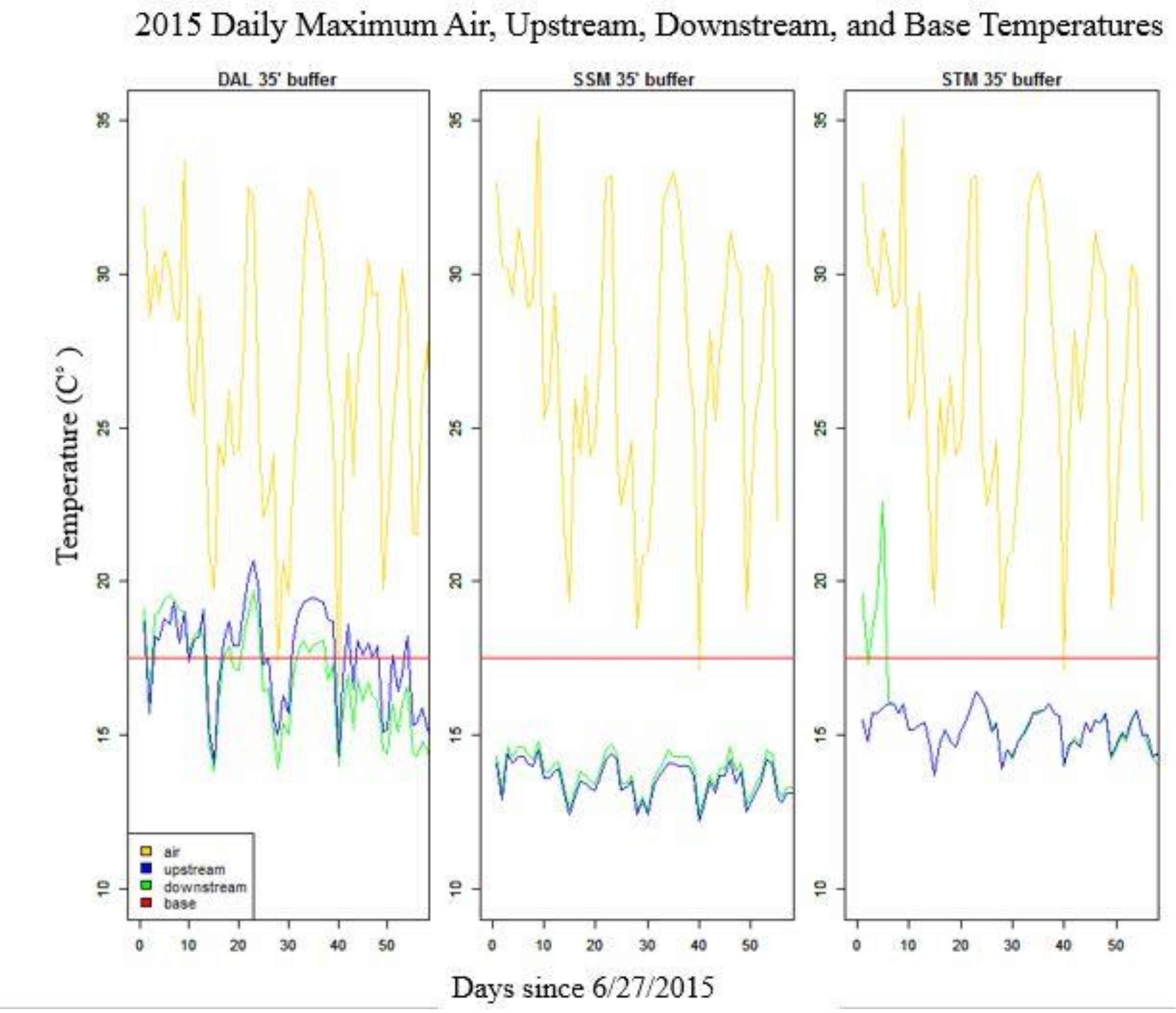



2014 Daily Maximum Air, Upstream, Downstream, and Base Temperatures
ELA 15' buffer
SSY15' buffer
DEG 15' buffer
VVA 15' buffer
VSA 15' buffer
STA 15' buffer
VPL 15' buffer
air
upstream
downstream
base
Temperature (C°)
10
15
20
25
30
35
0
10
20
30
40
50
Days since 6/27/2015
Daily Maximum Temperature
Time in Days
6/27/2014 to 8/21/2014

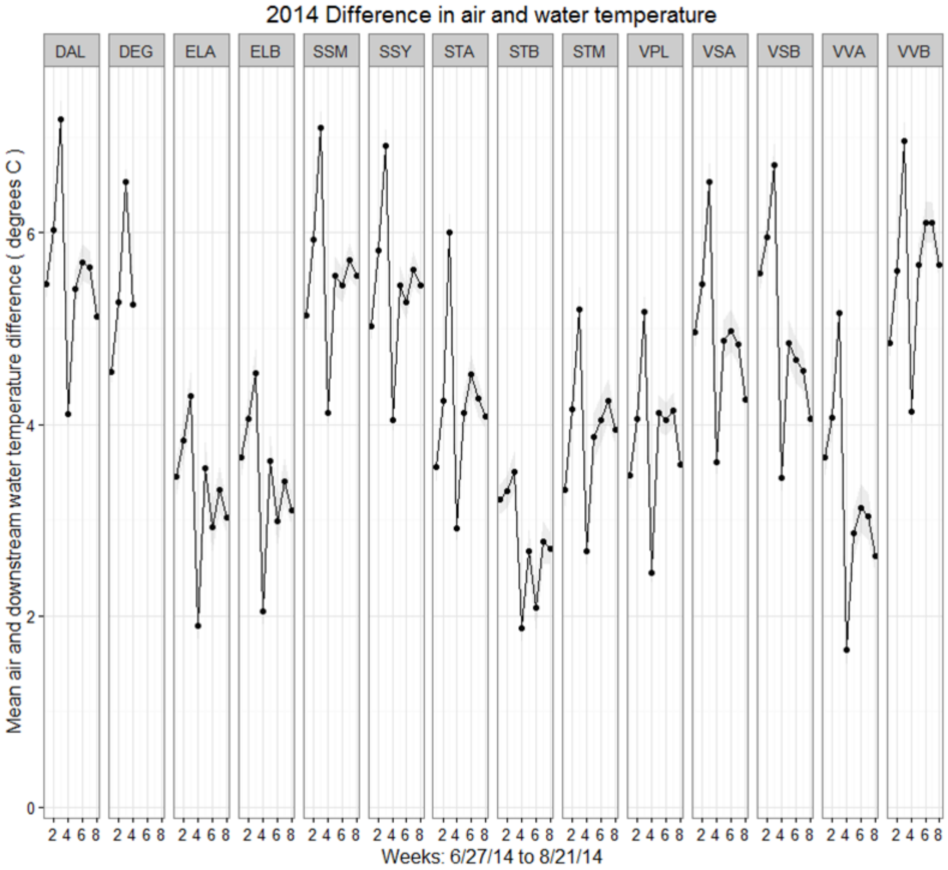

2014 Daily Maximum Air, Upstream, Downstream, and Base Temperatures
DAL 35' buffer
SSM 35' buffer
STM 35' buffer
Temperature (C°)
10
15
20
25
30
35
0
10
20
30
40
50
air
upstream
downstream
base
Time in Days
6/27/2014 to 8/21/2014

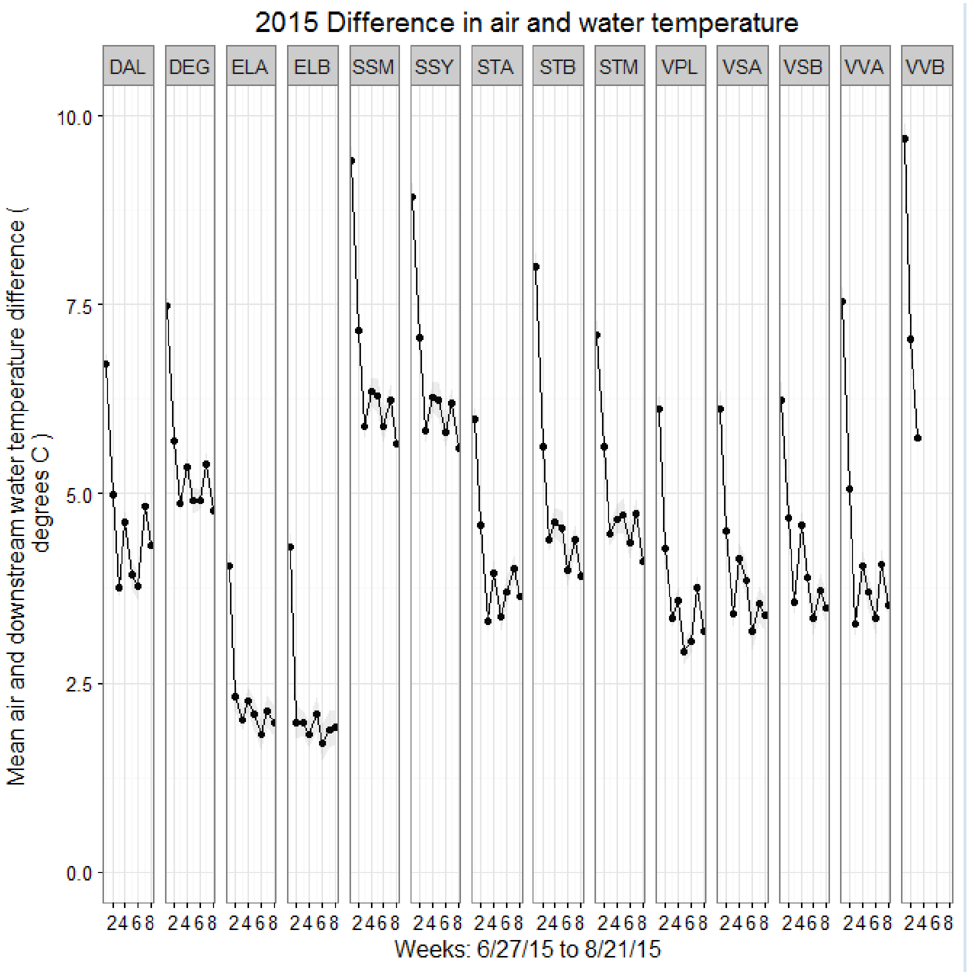

2015 Daily Maximum Air, Upstream, Downstream, and Base Temperatures
ELB 0' buffer
STB 0' buffer
VSB 0' buffer
VVB 0' buffer
air
upstream
downstream
base
Temperature (C°)
10
15
20
25
30
35
0
10
20
30
40
50
Time in Days
6/27/2015 to 8/21/2015


2015 Daily Maximum Air, Upstream, Downstream, and Base Temperatures
ELA 15' buffer
SSY15' buffer
DEG 15' buffer
VVA 15' buffer
VSA 15' buffer
STA 15' buffer
VPL 15' buffer
air
upstream
downstream
base
Temperature (C°)
10
15
20
25
30
35
0
10
20
30
40
50
Days since 6/27/2015
Daily Maximum Temperature
Time in Days
6/27/2015 to 8/21/2015

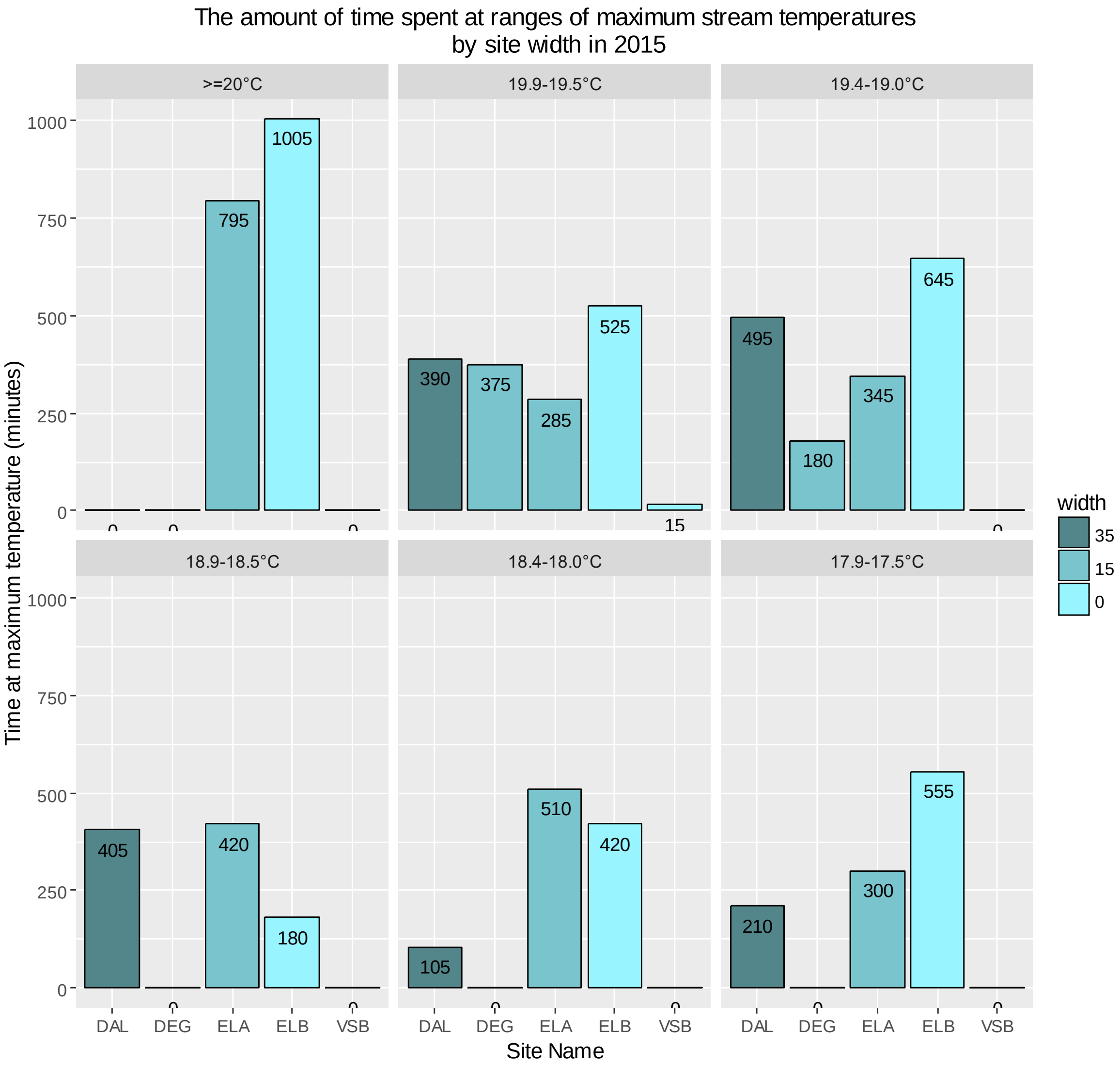

2015 Daily Maximum Air, Upstream, Downstream, and Base Temperatures
DAL 35' buffer
SSM 35' buffer
STM 35' buffer
Temperature (C°)
10
15
20
25
30
35
0
10
20
30
40
50
Days since 6/27/2015
🟨 air
🟦 upstream
🟩 downstream
🟥 base

Appendix V. Difference in Weekly Mean Air and Weekly Mean Water Temperature in 2014 and 2015.




2015 Difference in air and water temperature
DAL
DEG
ELA
ELB
SSM
SSY
STA
STB
STM
VPL
VSA
VSB
VVA
VVB
10.0
7.5
5.0
2.5
0.0
Mean air and downstream water temperature difference (degrees C)
2 4 6 8
Weeks: 6/27/15 to 8/21/15

Appendix VI. Amount of time spent at maximum stream temperatures for sites with daily maximums above 17.5 C



Appendix VII. Extension of Research

Public Outreach

Throughout the study period a number of presentations about the research project were delivered. Both professional and public audiences were targeted to facilitate conversation about current conservation and water quality mitigation practices. The end result was reaching 263 people at nine speaking events and presentations. Conclusions from the literature review for this study were incorporated into a response to the WA Department of Ecology’s draft of changes to the CAFO permit (winter 2015). It is likely that policy makers took these recommendations into account before solidifying the changes to the permit.

Table 1. A descriptive list of presentations delivered and the number of people who attended for the entire project period: 2014-2016.

Year	Presentation Description	Number of attendants
2014	Field Day at Sundstrom's Farm	5
2015	Salmon Recovery Conference, Vancouver, WA.- speaking presentation	40
	Master Gardener Booth Bellingham farmer's market- poster presentation	5
	Green Drinks meeting for environmental professionals, Bellingham, WA- poster presentation	50
	Focus on Farming Conference, Snohomish County, WA- speaking presentation	45
	Field day with Whatcom Conservation District-poster presentation	13
	Recommendation to WA Department of Ecology on CAFO policy	
2016	Regional Meeting of The Wildlife Society – speaking presentation	75
	Dairy Speaker Series at Whatcom Conservation District- speaking presentation	~35-40