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IMPACTS OF SMALL, SURFACE-RELEASE DAMS ON STREAM TEMPERATURE AND DISSOLVED OXYGEN IN MASSACHUSETTS

A Thesis Presented

by

PETER A. ZAIDEL

Submitted to the Graduate School of the
University of Massachusetts Amherst in partial fulfillment
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IMPACTS OF SMALL, SURFACE-RELEASE DAMS ON STREAM TEMPERATURE AND DISSOLVED OXYGEN IN MASSACHUSETTS

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ABSTRACT

IMPACTS OF SMALL, SURFACE-RELEASE DAMS ON STREAM TEMPERATURE AND DISSOLVED OXYGEN IN MASSACHUSETTS

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Dams fragment streams and rivers, with >14,000 in New England alone, and have the potential to significantly alter the physical, chemical, and biological characteristics of lotic systems. For example, dams can alter temperature and dissolved oxygen (DO) regimes, which can, in turn, affect species distributions, whole system metabolism, and nutrient processing rates. Moreover, changes in temperature signal life history cues (e.g., emergence, egg-hatching, migration) for many species of aquatic organisms, and present another avenue for dams to alter biotic communities. Despite the prevalence of small dams in the landscape and their potential significant impacts on temperature and DO, dams have not been well-studied and published impacts vary widely across sites. Given the variation in impact, I sought to quantify the impacts of small dams to stream temperature and DO, and to determine the drivers of inter- and intra-site variation in response. To accomplish this, I deployed 160 continuous temperature data loggers at 30 small, surface-release dams in Massachusetts. The majority of sites (61%) had higher temperatures downstream of the dam compared to upstream and most (85%) experienced decreasing temperatures with increasing distance downstream of the dam, such that the warmest temperatures were located closest to the dam. At approximately half of the temperature sites, flow had a homogenizing effect on temperatures throughout the study reach, whereby impacts were more pronounced (e.g., more warming, faster decay rates) under periods of low flow than under high flow conditions. Magnitude of warming varied greatly among sites, and this variation was explained best by landscape position and reservoir volume, with dams in smaller watersheds and with larger reservoir volumes experiencing greater warming magnitudes. Forest cover, dam height, and the presence of an auxiliary spillway best predicted the downstream temperature decay rate, with temperatures cooling fastest downstream of shorter dams in forested basins that did not have an auxiliary spillway. I used continuous DO loggers upstream, within the impoundment, and downstream of 12 dams to identify dam impacts to DO. Most sites experienced lower DO (66%) within the impoundment compared to upstream; however, 58% of the sites showed no difference in diel ranges between these reaches. The effect of dams on downstream DO was mixed, with increases, no change, and decreases relative to upstream condition; however, the majority of sites (58%) experienced a suppressed downstream diel range relative to upstream. The upstream slope, basin size, and dam height drove the impoundment response, such that dams with steeper upstream reach slopes, located in smaller basins, and with shorter dam heights experienced the greatest decreases in impoundment DO relative to upstream. Differences between downstream and upstream DO were best explained by upstream slope and impoundment volume, whereby sites with steeper upstream reaches and larger volumes of water within the impoundment experienced the largest decreases in downstream DO when compared to upstream reaches. These results may help managers prioritize dam removal at sites where a dam is having larger and more negative (e.g., elevated temperatures, decreased DO) impacts, and therefore where the greatest benefits should occur following restoration.

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CHAPTER 1

INTRODUCTION

1.1 Damming Throughout History

Humans have built dams for millennia to tame streams and use them to benefit society by providing drinking water, as a source of cheap and easy power, and to control flood waters (Poff and Hart 2002). The practice of damming rivers was so ingrained in European culture that settlers to the New World would typically immediately build a dam (Bishop 1861, Reynolds 1983). Indeed, dams were fundamental to the success of early European settlers. In the northeast U.S., dams were critical in leading the industrialization of the country, as rivers powered grist, textile, and paper mills (Walter and Merritts 2008). The U.S. continued to build dams at an ever-increasing rate until its peak in the decades following the Great Depression and World War II (1950 – 1970), after which the construction of new dams in the country effectively came to a halt (Graf 1999, Poff and Hart 2002, FEMA 2012). For over 300 years, from the mid-1600s when the first known dam was believed to have been built in the U.S. through the 1970s, dams in America, both through their construction and through the mills and hydropower facilities they powered, were equated with economic opportunity and social benefits (Walter and Merritts 2008).

1.2 Ecological Impacts of Dams

Although dams have provided great benefits to society, they have come at great costs. Dams are physical, static barriers within a dynamic river system with a multitude of wide-ranging ecological impacts. The most fundamental impact of a dam is the conversion of a flowing stream into a pond. The magnitude of this change can vary between different types of dams, with run-of-river dams generally having a smaller impounding effect on a stream than a larger managed dam, such as a flood control or hydropower dam. A major impact of damming and the conversion of the stream into a pond is the trapping of sediment behind the dam. As water carrying suspended sediment enters the still water of the impoundment, particles drop out of suspension and settle within the impoundment (Petts 1980, Poff et al. 1997, Stanley and Doyle 2003). Contaminants such as PCBs or PAHs can easily bind to these fine particles, and dams on rivers with a history of industry may be susceptible to the accumulation of high levels of these contaminants. The collection of fine sediment upstream of dams leads to "sediment starved" downstream reaches. Reduced fines can lead to high rates of erosion in downstream areas with fewer riparian plants and increased scour of the stream bed relative to upstream reaches (Kondolf 1997). In addition to the water quality issues posed by elevated contaminant levels within the impoundment, dams can impact stream temperature and dissolved oxygen (DO) levels, with varying directionalities and magnitudes of impact across dams. Bottom-release dams are generally tall structures with deep impoundments that release unnaturally cool water in the summer and warm water in the winter from the ~4 °C hypolimnion layer of the deep impoundment formed behind the dam (Holden 1979, Ward and Stanford 1979, Armitage 1984). The hypolimnion of these impoundments is low in oxygen, and generally these structures release oxygen-poor water to downstream reaches (Bednarek 2001, Bednarek and Hart 2005). The impacts of surface-release dams on temperature and oxygen are less consistent than those of bottom-release dams. Reported results on the effects to downstream temperature include warming, no change, and, rarely, cooling relative to upstream conditions (Bushaw-Newton et al. 2002, Lessard and Hayes 2003, Singer and Gangloff 2011, Kornis et al. 2015). The impacts to DO are also variable amongst sites with reported results ranging from decreased downstream DO, to no discernable difference between upstream and downstream concentrations, to increases in downstream DO (Bushaw-Newton et al. 2002, Lessard and Hayes 2003, Maxted et al. 2005, Ignatius and Rasmussen 2016). The variation in water quality response across these studies and lack of tested drivers of this variation make it difficult to predict how a small dam, and consequently its removal, will affect temperature and oxygen.

The ecological impacts of dams are not limited to abiotic stream features, as dams have also been shown to impact fish, macroinvertebrate, and plant communities within a stream ecosystem. Arguably the most well-known biotic impact is the blocking of upstream-migrating anadromous fish. Dams have decimated historically abundant fish runs in coastal watersheds, with examples of losses in Atlantic salmon (*Salmo salar*), American shad (*Alosa sapidissima*), alewives (*Alosa pseudoharengus*), blueback herring (*Alosa aestivalis*), and American eel (*Anguilla rostrata*) in New England (Limburg and Waldman 2009, Hall et al. 2011, Hitt et al. 2012, Mattocks et al. 2017), and decreases of species along the west coast of the U.S. such as Pacific salmon (*Oncorhynchus* spp.) (Raymond 1979, Nehlsen et al. 1991, Kareiva 2000). Dams can also block the upstream- and downstream-movements of resident riverine species (Santucci et al. 2005), which over time can isolate populations and lead to an elevated extinction risk of those populations (Dunham et al. 1997, Letcher et al. 2007). In cases where a dam has a significant effect on downstream temperatures, there can be a subsequent shift in fish species composition and change in species richness immediately downstream of dams. For instance, dams that warm a coldwater habitat can lead to decreased native coldwater species downstream of the dam (Lessard and Hayes 2003, Bellucci et al. 2011). Benthic macroinvertebrates are impacted primarily due to changes in habitat resulting from damming. The impoundment, with its slow-moving water and fine sediment deposition, is often comprised predominately of oligochaetes and chironomids (Santucci et al. 2005). Downstream reaches may have decreased EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa and elevated chironomids relative to upstream reaches (Bushaw-Newton et al. 2002), while changes in food characteristics may have been the driver of decreased abundances of shredders and collector-gatherers downstream of a dam (Martinez et al. 2013). Plant communities may also be impacted by damming, as the initial impounding of water and subsequent widening of the wetted channel kills trees and riparian plants not suited for submersion. The reduced flows, elevated nutrients, and warmer temperatures of the impoundment can be highly conducive to macrophyte and algal growth (Soballe and Kimmel 1987), which may put impoundments at a higher risk for harmful algal blooms than free-flowing reaches (Przytulska et al. 2017). Downstream of the dam, reduced sediments and nutrients can result in decreased rooting habitat and a reduced macrophyte communities.

1.3 Financial Costs of Dams

In addition to numerous ecological costs, dam owners are subject to a number of financial costs. There are fees simply to operate the dam, ensure its safety, repair structural deficiencies, insure the dam, and create a plan in case of dam failure. Based on their height, volume of impounded water, and location with respect to other human infrastructure, dams are rated on their hazard potential in the event of a failure. There are three hazard classes: high hazard dams are those in which failure would result in near-certain loss of human life or property; significant hazard dams are those where failure would likely have large impacts to downstream communities and infrastructure, but not loss of life; and low hazard dams, if they were to fail, would likely have minimal impacts to downstream infrastructure (MDCR 2005, USACE 2016). To ensure the structural integrity of a dam, dam owners must hire an engineer to inspect the dam, with the frequency of this inspection related to the dam's hazard class. High hazard dams must be inspected every two years, significant hazard dams every five years, and low hazard dams should be inspected every 10 years (MDCR 2005). The cost of these inspections is on average $5,000 per dam (DeNucci 2011). An audit in Massachusetts by the State Auditor's Division of Local Mandates reviewing the economic impact of Massachusetts dam safety laws states that developing an operating manual that details best practices for operation of the dam is a one-time cost of ~$5,000, and a number of municipally-owned dams would require ~$5,000 per year in recurring costs to operate their dam up to code (DeNucci 2011). Massachusetts has 627 municipally-owned high or significant hazard dams, of with 75% do not have an Emergency Action Plan (costing on average $10,700 per dam to create), which details the actions dam operators and downstream communities would take in response to a failure of the dam. The average cost to repair the 96 municipally-owned, high and significant hazard dams in Massachusetts in need of repair is $573,154 per dam (DeNucci 2011). In addition to these fees, dam owners must pay annual fees to insure their dam. Collectively, these costs can result in the dam becoming economically unfeasible for dam owners to maintain.

The argument is often made that smaller, non-revenue-generating dams are more economically sensible to remove, especially as they age (FE/AR/TU 1999). While this may be true, a recent study has found that those budgeting the construction of a large dam consistently under-estimate the costs, often negating to consider inflation or debt servicing (Ansar et al. 2014). This trend has caused the construction costs of most large dams to run significantly (56%) over budget (WCD 2000). As such, it seems that while small dams may make more economic sense to remove, even large revenue-generating dams may not be as economically practical as previously thought.

1.4 Shifting Tides: Dam Removal

There are several ways to mitigate against the ecological and economic impacts of dams. Partial solutions to the problem can include building fish ladders, dredging impoundment sediments, or repairing a dam in poor condition. However, the most effective solution, that addresses the root of the impact most completely, is to remove the dam. Several examples from the early 1900s provide evidence that the practice is not a new means of eliminating the ecological and economic costs of dams. In 1839, while paddling the Concord and Merrimack Rivers in Massachusetts, Henry David Thoreau recognized the negative effect of dams on local fish populations and pondered how the dam would fare against a crow bar (Thoreau 1849). After being weakened by flood waters, the 50-foot-tall Hartford Manufacturing Company Dam in Glastonbury, CT was removed in 1904 to reduce the safety hazard of the structure (Barber 1990, Hubbard and the Historical Society of Glastonbury 2012). The Sunbeam Dam, the only dam on the mainstem Salmon River in Idaho, was removed in the early 1930s, after it was abandoned by a defunct mining company, solely for the purpose of restoring the river's salmon runs

(Pohl 2002, Hawley 2011). These examples hint at what Pohl (2002) found were the three most commonly cited reasons for removing a dam: improving ecological conditions, removing safety hazards, or eliminating maintenance costs. While these examples demonstrate that dam removal is not a new concept, they were unique for their time, and it was not until the 1980s that the idea of removing dams became a commonly-accepted restoration practice and a more common socially acceptable ending to a dam's story.

There has been an exponential increase in the number of American dam removals that began in the late 20th Century (Bellmore et al. 2017) and has continued to current. There are several possible explanations for this increase, one of which is that the country predominately stopped building dams in the late 1970s, and it was not until the construction of dams slowed that the negative effects of dams were recognized (Graf 1999). These ecological effects were being realized at a time when American society was placing higher value on ecological systems (Pohl 2002). Another explanation involves the age of American dams; as Dr. David Montgomery states in the popular documentary *DamNation*, "like all constructed things, dams have a finite lifetime" (Knight et al. 2015). Unfortunately, several of those finite lifetimes came to end with failure that resulted in human casualties in the 1970s (e.g., Teton Dam in 1976 and Laurel Run Dam in 1977 killed 11 and 40 people, respectively) that shed light on the safety hazard posed by this infrastructure (Rose 2013, "Teton Dam History" 2016). Large dams built in the 1930s at the beginning of the nation's "golden era of dam building" reached the end of their 50-year design and permitted life in the 1980s (FE/AR/TU 1999). In 2000, the World Commission on Dams determined that the benefits of large dams rarely exceed their ecological and social costs (WCD 2000), calling an economic reason for their removal

(Pohl 2002). As the economic, safety, and ecological costs of dams were being better understood and realized, the late 20th Century marked a turning point in the nation's attitudes towards dams. The U.S. now consistently records >50 dams removed per year, with the total removed exceeding 1,300 through 2016 (American Rivers 2017).

1.5 Ecological Responses to Dam Removal

As more dams are removed, our understanding of how streams respond to removal continues to improve, particularly with regards to geomorphic and biotic responses (Bellmore et al. 2017). Some of the most dramatic changes following dam removal are geomorphic changes that respond predictably based on the removal strategy and impounded grain size. Dams removed all at once generally have an upstream-moving knickpoint that headcuts through the formerly-impounded sediments, while those dams removed in stages generally have a downstream-extension of the reservoir delta, a term referred to as progradation (Grant and Lewis 2015). Regarding sediment size, cohesive, fine sediments generally erode at a slower rate than non-cohesive, smaller particle sizes (Grant and Lewis 2015). In several studies of sediment flushing dynamics of dam removals along the Atlantic coastline, researchers identified a two-phase exponential pattern of sediment erosion rates following dam removal. In this two-stage pattern, roughly half of the reservoir sediments are flushed with the initial drawdown ('process-driven') to base level, at which point the remaining 50% of sediment will be flushed only following large, bank-topping floods ('event-driven') (Pearson et al. 2011, Sawaske and Freyburg 2012, Collins et al. 2017).

Similar to geomorphic responses, biotic responses to dam removal have also been fairly well-studied. Many studies have demonstrated positive impacts of dam removal on fishes, particularly anadromous fishes. Despite being cut off from these upstream, historic spawning grounds for centuries, anadromous fish return to these reaches and utilize the newly-opened spawning habitat quickly (Hogg et al. 2013, Pess et al. 2014, Magilligan et al. 2016b). Beyond the simple elimination of a barrier allowing access to upstream reaches, changes in the abiotic characteristics of a stream following dam removal can also facilitate positive biotic responses. For instance, the downstream aggradation of fine sediments that eroded out of a former impoundment following dam removal re-established suitable spawning habitat for native sea lamprey (*Petromyzon marinus*) in a Massachusetts stream (Magilligan et al. 2016b). Given the dependence on gravel substrate amongst other fish species such as salmonids (Opperman et al. 2005), geomorphic changes above and below a former dam site should be expected to have benefits to numerous other biota. This dependence on abiotic conditions and geomorphic changes brought about by dam removal is not limited to fish, as several studies have observed a shift from a chironomid- and oligochaete-dominated macroinvertebrate community within an impoundment to a more typical riverine assemblage dominated by EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa with less chironomids and oligochaetes following dam removal (Bushaw-Newton et al. 2002, Stanley et al. 2002). This shift is likely due to changes in habitat (e.g., substrate size, flow rates, water depth) as the former reservoir returns to a natural riffle state (Barbour et al. 1999).

Following dam removal, the drop in the water table may desiccate riparian plants that were previously submerged in the former impoundment (Shafroth et al. 2002), though these changes appear to be restricted to areas that experienced the greatest geomorphic and hydrologic changes (Lisius et al. 2018). The dewatered impoundment sediments, which reflect both a recently disturbed habitat and often contain decades of nutrient build up, become a newly-available and attractive habitat for invasive plant species. Invasion by nonnative plant species such as reed canary grass (*Phalaris arundinacea*), common reed (*Phragmites australis*), and purple loosestrife (*Lythrum salicaria*) is one of the seven most common management concerns identified by Tullos et al. (2016). While studies have observed invasion by these species in the former impoundment (Lenhart 2000, Orr et al. 2006), others have shown a lack of invasion by nonnative plants following dam removal (Lisius et al. 2018).

Water quality changes, by contrast, are some of the least studied aspects of dam removal (Bellmore et al. 2017). The results that have been published suggest variable responses to dam removal with both reported improvements (return to more natural conditions) and examples of no change. The sites that experience benefits following removal are those in which there were negative effects of the dam (Tuckerman and Zawiski 2007, Muehlbauer et al. 2009, Kornis et al. 2014). In contrast, sites that experience no change following removal are typically those in which there was no effect of the dam (Bushaw-Newton et al. 2002, Orr et al. 2006). Ultimately, these studies highlight that the benefits of dam removal should only be expected in sites that are negatively impacted by the dam.

1.6 Financial Benefits of Dam Removal

One of the largest financial savings related to dam removal is that removal is a one-time cost, which stands in sharp contrast to the recurring costs of dam maintenance, inspection fees, insurance, and costs to keep the dam in compliance with current regulations that persist as long as the dam is in the river. Bringing a noncompliant dam up to environmental code (e.g., building a fish ladder) can be 3-5.5x more expensive than removal (Strassman 2011). Several studies have performed cost-benefit analyses on dam removal, and interestingly, both found the costs of dam removal to be about 60% of the costs of maintaining the structure (FE/AR/TU 1999, MDFG 2015). Moreover, financial help (e.g., grants from state and federal agencies and NGOs) is often available to offset the cost of dam removal; financial assistance to help maintain a dam, on the other hand, is less common. In the case of the two privately-owned dams in a report published by the Massachusetts Department of Fish and Game (MDFG)'s Division of Ecological Restoration in 2015, the dam owners paid 0.5% and 3% of dam removal total costs, which in 2014 dollars was $11,000 and $31,000, respectively (MDFG 2015). Beyond the physical costs of the dam, property values for both the dam owner and neighbors may increase with loss of flooding potential (Lewis et al. 2008, Provencher et al. 2008, MDFG 2015). While more difficult to quantify, removing a high- or significant-hazard dam can also eliminate large potential payouts by the dam owner to local businesses and for emergency response in the event of a dam failure (MDFG 2015).

1.7 Variation in Dam Removal Among States

Despite the documented benefits of dam removal, there is a wide variation in the number of dams removed within each state (Pohl 2002, Bellmore et al. 2017). While as a nation the U.S. has removed over 1,300 dams through 2016, 40% of these removals have occurred within just three states; Pennsylvania, Wisconsin, and California have removed 299, 135, and 99 dams, respectively (data from American Rivers 2017, Bellmore et al. 2017). At the opposite end of the spectrum, neither Mississippi nor Oklahoma have a single reported dam removal. Most states (70% according to Bellmore et al. 2017) have removed between 1 – 25 dams. Looking broadly at patterns of dam removal amongst U.S. states, those along both coasts and in the upper Midwest remove more dams than those in the southwest, intermountain west, and lower Midwest (Bellmore et al. 2017).

While the aforementioned high number of removals within states such as Pennsylvania and Wisconsin may be attributed predominately to particular pieces of dam-removal supporting and funding legislation, broader patterns of removal may be driven by social and political attributes of individual states. I tested whether the number of dams removed within a state was related to the number of dams within a state, the density of dams within the state (number per area), the state's gross domestic product (GDP), the annual rainfall within the state, and the state's political party affiliation (Table 1.1). I hypothesized a positive relationship between the number of dams removed within a state (response variable) and the number of dams within the state, the density of dams within the state (dams/area), the state's GDP, and the average annual rainfall within a state (predictor variables). More dams in a state may mean that there are more older, failing dams within a state, while states with higher GDP may have more resources to remove dams. Additionally, states with more rainfall or a higher density of dams (more dams/area) may be less reliant on dams in general (rainfall), or individual dams (dam density), for water storage (Table 1.1). Given potential differences in spending between democratic and republican states on environmental issues (Jones and Dunlap 1992, Elliott et al. 1997), I hypothesized that democratic states would remove more dams than republican states (political affiliation determined by voting for the same party at least 3 of the past 4 presidential elections) (Table 1.1). Florida, Iowa, and Ohio were not included as they were split 2–2 between the past 4 elections. I log-transformed continuous variables (number of dams, number of dam removals, and GDP) to reduce skew and meet assumptions of normality.

Two of the five tested variables significantly predicted the number of dam removals within a state. A state's political party affiliation as determined through the past four presidential elections significantly ($p< 0.001$) predicted the number of dams removed within a state ($R^2= 0.24$) as hypothesized, with democratic states removing more dams than republican states. Democratic and republican states generally exhibit differences in spending on environmental issues (Jones and Dunlap 1992, Elliott et al. 1997), and these results suggest that these differences may extend to river restoration efforts in the form of dam removal. As hypothesized, a state's GDP also positively predicted ($p= 0.001$) the number of dams removed within the state ($r^2= 0.21$). Given that trends in the economy have been suggested as drivers in inter-annual spending on conservation efforts (Pergams et al. 2004), the positive relationship observed offers support to the hypothesis that states with larger economies have the resources to remove more dams than those with smaller economies. Neither the number of dams within a state ($p= 0.23$), the density of dams within a state ($p= 0.51$), nor the average annual rainfall ($p= 0.52$) significantly predicted the number of dams that a state removed. Several studies (Graf 1999, Poff and Hart 2002, Magilligan et al. 2016a) have shown that the NID severely underestimates the number of dams in the country, and especially underestimates the number of small dams, which, given their size, are easier and cheaper to remove. This discrepancy may be behind the lack of a relationship between the number and density of dams within a state and the number of dams removed within a state. It was surprising that average rainfall did not come out as significant, though water resources may be better modeled as the storage/runoff ratio, as Graf (1999) calculated, rather than total precipitation.

Work done by Kahan et al. (2015) identified that for politicized topics, relationships between a continuous predictor (e.g., individual intelligence) and attitudes of risk perception regarding the politicized topic (e.g., the threat of global warming) exhibited a different and diverging relationship when individual's political leanings (i.e., liberal or conservative) were included, than when the continuous predictor was tested alone. Given that dam removal can be a highly politicized topic (Fox et al. 2016), it seemed reasonable that there may be a similar pattern of divergence in response when political affiliation was modeled with continuous predictors of state dam removals. As predicted, including an interactive term for political affiliation with continuous predictors changed the relationship among several of the variables. A state's GDP still had a significant positive relationship on the number of dams removed for democratic states ($p= 0.03$), but did not for republican states ($p= 0.79$). This result (model $R^2= 0.42$) sheds a more nuanced light on the relationship with GDP identified above, whereby only democratic states see more dam removals with larger state economy sizes. Just as global warming is viewed as a greater threat with increasing intelligence among liberals, but viewed as less of a threat with increasing intelligence among conservatives (Kahan et al. 2015), democratic states may view dams as more of a threat than republican states and are limited by budgets in the number of dams being removed within a state. Interestingly, the number of dams within a state, a variable not significant on its own, also exhibited this pattern and in fact best predicted the number of dams removed within a state ($R^2= 0.54$; Figure 1.1). For republican states, the number of dam removals within a state was not related to the number of dams located within that state (p= 0.74); however, for democratic states, the number of dams in each state was a strong predictor for the number of dams that state has removed (p< 0.001). If there are more dams in a state, there may be more widespread and recognized negative impacts to ecosystems from those dams that a democratic state may be more concerned over and wish to eliminate than a republican state. Additionally, there simply may be more failing dams in states with more dams, posing greater threats to infrastructure (e.g., to hospitals and schools) that democratic states again may be more likely to view as a risk that should be addressed via removal.

There are, however, some potential limitations or drawbacks to these results that certainly warrant further investigation. One of the largest potential issues is the bias in the data source used for the number of dams within a state – the NID. To keep the database manageable, the NID has restrictions for inclusion based on size (height and volume of a dam/reservoir) and hazard class (high and significant hazard dams are included; USACE 2016). There may be an underlying pattern whereby a number of northeastern U.S. states, the majority of which are democratic and some of the first settled, may have dams listed in the NID because of a hazard classification resulting from the age and subsequent poor condition of the structures. Republican states may predominately have dams included in the NID due to size and storage reasons. Using dam totals from state agencies, the ages of dams, or involving the hazard classes of dams in these models may help to uncover some of these potentially hidden drivers behind these results.

1.8 Motivating More Dam Removals

One potential explanation for differences in the number of dams removed per state is managers' concerns about uncertainties regarding how local resources will be impacted by dam removal. One of the more high-profile examples of negative effects of dam removal occurred on New York's Hudson River, when polychlorinated-biphenyl (PCB)-contaminated sediment was released downstream with the removal of the Fort Edward Dam in 1973. Bellmore et al. (2017) speculated that this dam removal-induced contaminant release may have caused a state-wide attitude of speculation of or even opposition towards dam removal as a restoration tool. Interestingly, despite this possible state-level effect, the downstream release of contaminated sediments was not listed as one of the seven most common management concerns related to dam removal, as summarized by Tullos et al. (2016), though this may be because sediment testing is now performed prior to most dam removals. Broadly, these seven most common concerns involved geomorphic, biotic, and infrastructure impacts and changes to reaches upstream of, within, and downstream of the former impoundment. Geomorphic concerns and changes within the former impoundment were most common. Related to geomorphology, managers specifically were concerned with how much and how quickly impoundment sediment will erode, channel incision (and possible infrastructure and biotic impacts) upstream of the dam, sediment deposition and accumulation downstream of the dam, and decreased downstream water clarity resulting from high suspended sediment loads. Two of the seven concerns involved the spread of nonnative species into the former reservoir, with managers reluctant to remove dams if there is potential for nonnative plants to establish in the formerly impounded sediments or for nonnative fish communities to expand upstream of the former dam. While dam removals can impact infrastructure, only one of the seven concerns—how the reservoir drawdown would affect local water infrastructure using water directly from the reservoir or from elevated water tables near the reservoir—directly revolved around infrastructure (Tullos et al. 2016). While these concerns arise from very real possible negative impacts of dam removal, advances in the collective understanding and a consideration of local conditions can assuage a number of these concerns. For instance, if the downstream reach of a dam has no known instances of nonnative fish, upstream invasion following removal should not be a concern. Dam removal deconstruction can be designed to alleviate some concerns. For example, phased removals, whereby the dam is not removed all at once but in a series of partial removals, display slower rates of initial impoundment erosional rates than do instantaneous removals (Wilcox et al. 2014, Randle et al. 2015, Major et al. 2017). Management concerns over possible negative ecosystem and infrastructure responses to dam removal can lead to a project not being pursued or to delays in the timing as concerns are addressed. Understanding the local landscape and determining which concerns are valid for a given proposed project can help focus efforts and allow managers to work and move more efficiently through a dam removal project.

In addition to addressing concerns, dam removal practitioners and legislatures can create incentives and structure at the state level to help increase the number of dam removals occurring within a state. Several reports have investigated similarities of states that have been successful at removing dams, and their shared findings highlight the need for a dedicated team of practitioners working together in a legislative environment that supports dam removal (Lindloff and Wildman 2006, Zinder et al. 2009). States that are successful at removing dams not only have dam safety laws, but also actually enforce those laws. Successful states also have a dedicated funding source for dam removal projects. Third, states with a large number of successful dam removals understand the importance of outreach and engagement, both in the form of educating the public on the benefits of dam removal (Zinder et al. 2009) and providing information to dam owners about the permitting process (Lindloff and Wildman 2006). The fourth shared feature of a successful state was teamwork between practitioners, collectively including both the need for different state and federal agencies and non-governmental organizations to work together (Zinder et al. 2009) and a forum via which practitioners can share lessons learned with others (Lindloff and Wildman 2006). For states wishing to increase the number of dam removals within their boundaries, tackling these common features will be helpful in achieving the goal of more dam removals. Some of these (e.g., changing legislation to support enforcement of dam safety laws) may be a slow process, but others (e.g., creating outreach documents and establishing forums for practitioners to share success stories and lessons from failures with one another) should be easy first steps in working towards an increased number of dam removals. Ultimately, states need dedicated staff working for both governmental and non-governmental organizations collaboratively towards a common goal of restoring rivers via dam removal.

1.9 Local Community Dimensions of Dam Removal: Challenges and Opportunities

Despite the shifting tide towards increased numbers of U.S. dam removals, local opposition still exists in earnest in many situations. One of the most fundamental barriers to dam removal is a lack of trust between locals and practitioners. The practitioners often are employees of state agencies, federal agencies, and national non-governmental organizations, and as such can be viewed as outsiders by local townspeople (Williams and Stewart 1998, Fox et al. 2016, Magilligan et al. 2017). In extreme cases, these negative attitudes can extend into class conflict, where locals are immediately made to feel like the poor, dumb, country bumpkins whose community is being ripped apart by the rich, elite urbanites of some governmental offices (Fox et al. 2016). This “us vs. them” attitude can be particularly problematic for dam removals that do not make a good first impression on locals, as Magilligan et al. (2017) point to several instances where community members were immediately turned off by what they perceived to be practitioners walking into town with a decision already made regarding the future of the dam in question. In this situation, practitioners can be easily painted in a light not congruent with the town’s image and vision for itself.

Dams have been around for a long time, and especially in New England, have been focal points of communities for several hundred years. Many towns were created around mill dams and prospered from the jobs and revenue that came with damming a river (Hunter 1979, Steinberg 1991). This places the dam not just as a feature of the physical landscape of the community, but as a feature at the core of the historical and cultural identity of the community. Where a dam is central to a community’s self-image, an attempt to remove the dam becomes an attempt to erase part of the community’s identity. As many dams are several hundred years old, the dams and the landscape features they create (i.e., the pond behind the dam) have been a part of the collective memory for generations (Fox et al. 2016). These longstanding features of the landscape have the potential to create “ecstatic memories” that are most intensely remembered and ingrained within an individual’s memory (Chawla 1990, Gieseking et al. 2014). Support for this idea is evidenced by one community member opposed to the removal of the Bondsville Dam on the Swift River in Belchertown, MA who went so far as to say, “you kill the dam, you are killing a part of me” (Fox et al. 2016). When a community’s place attachment to a dam becomes that strong, community opposition is a major barrier to removal.

This place attachment is problematic because the place townspeople are attaching to is an artificial one; it is manmade, and not designed to exist in perpetuity. As pieces of infrastructure, dams require maintenance to remain intact structures over time, and as discussed above, this maintenance can be expensive, and much more than the cost of dam removal (FE/AR/TU 1999, MDFG 2015). By opting to pay more to maintain the structure than it would cost to remove it, locals are placing a higher value on the dam than would be economically responsible. This high value of the dam may partly explain why conflicts surrounding dam removal are more often considered to be “value conflicts”, in which the question is not which restoration technique is best suited to meet the needs of the project, but if the restoration should be taking place at all (Fox et al. 2016). This stands in sharp contrast to many other types of river restoration (e.g., bank stabilization, natural channel design), where conflicts could be considered “interest conflicts” (Lord 1979), in which debates surround the mechanisms of restoration and whether or not those mechanisms will achieve the desired goals of the project.

A key component to successfully removing a dam is establishing trust between practitioners and the local community. An easy and simple way to establish trust is to designate someone early on in the project as the “local champion”. The term has been used across a number of disciplines – from public health to road construction to issues of conservation (Pentz 2000, Slotterback 2010, Young et al. 2016), and refers to a member of the community who is well-liked and well-respected, and who can help ease potential tensions between practitioners and locals and serve as a mediator between the two parties.

A local champion can be crucial in coordinating outreach and mediating negative attitudes such that locals feel the project is a partnership and not a form of governmental overreach. One of the greatest benefits this can afford is open dialogue between the community and those wishing to remove the dam. Successful outreach and dialogue with community members may be able to shift the value conflict discussed above to an interest conflict, in which all stakeholders agree that the action of removing the dam would be best for the community, and discussions may then shift to the best means to accomplish this action. While certainly not a guarantee to any project's success, open communication and a willingness to compromise from both proponents and opponents to dam removal should be a primary focus.

While serving as a barrier in some cases, place attachment and local pride in a community's past may actually encourage dam removal and river restoration efforts. McClenachan et al. (2015) share examples from Maine, in which coastal communities have been able to rally around their historical legacies of abundant anadromous fish runs to support dam removal. These communities, which include Native American tribes, have found that taking pride in their once-abundant alewife runs can enact a positive feedback cycle between restoration of streams, fisheries, and local economics. Here, townspeople are still connecting to their local history, but they are attaching to and taking pride in their ecological past over their industrial past. This may be an easier sell in coastal communities, whose memories of former economic glory days are directly dependent on healthy aquatic ecosystems (ASMFC 2009, Hall et al. 2012), then it would be in inland, mill communities whose economic nostalgia had little reliance on ecological health. It also may be easier for coastal communities to get behind dam removal (with their goals of restoring anadromous fish runs) given their proximity to the ocean. This landscape position often means that the dam in their community may be the first barrier to migrating fish and removing it may have a more immediately obvious impact on their local fisheries than upland dam removals. The increasing number of lowland dam removals though mean that anadromous fish are swimming further upstream today than they have in decades (Crane 2009, Hitt et al. 2012). Upland dams previously cut off from returning anadromous fishes may then serve as the first upstream barrier to these fish, and it is possible that an upstream-moving domino effect of dam removal and returning fish could help to spur upland removals, as seen on the Kennebec River in Maine with the removal of the upland Fort Halifax Dam 9 years after the removal of the downstream Edwards Dam (Crane 2009). It may then become possible, with the upstream-moving domino effect, to reshape community's sense of place, with returning fish serving as a visual reminder of an even older version of the community's history. In such a way, just as more salmon (returning to spawn and as sources of marine-derived nutrients) lead to more salmon (in the form of more productive systems to support larger juvenile populations; Nislow et al. 2004, Williams et al. 2009), more dam removals (in lowland, coastal streams) may lead to more dam removals (in upland areas).

At the root of many of the conflicts between practitioners and townspeople may be the simple fact that people are naturally averse to change. As discussed above though, practitioners have tools to help the public see the value of change in their community. Establishing trust through the use of a local champion can be pivotal, as can the seemingly simple feature of approaching locals with humility and being open to dialogue. Knowing that communities may have potentially deep-seated and long-standing connections to a place (e.g., dam) that outsiders may only see as an artificial landscape in need of improving for ecological or economic purposes may help practitioners to understand the ground from which local opposition to dam removal grows. The idea of removing a key feature of a small town may seem radical to locals, and practitioners should discuss these perspectives with communities. Under these softer approaches, practitioners may find that local opposition to a removal is not an unwavering position, but rather a result of the community not being felt that its collective voice is heard (Fox et al. 2016).

1.10 Conclusion

With an estimated 2 million dams in the country (Graf 1993), there are many dams that are failing us ecologically, structurally, and economically. To again quote Dr. Montgomery, "it's not time to pull out every dam in the country; that would be economically foolish. But it would be just as foolish not to rethink every dam in the country, and try and decide which are the ones that actually still make sense in the 21st century" (Knight et al. 2015). In the following chapters, I hope to provide some assistance to the collective 'rethinking of dams in the country', by investigating what factors influence the impact of dams on stream temperature and DO, and thus which dams would provide the greatest water quality benefits to dam removal.

The second chapter of this thesis investigates the impacts of small, surface-release dams on stream temperature. Previous studies have reported a wide range of impacts to temperatures downstream of small, surface-release dams. Additionally, the few studies that investigated temperature over a distance downstream of these small dams did not quantify the rates at which temperature changed, and thus, were unable to determine the extent of thermal impact. As such, I aimed to quantify the downstream thermal effects of small dams, measuring the magnitude of warming, the rate of temperature change downstream of the dam, and the distance downstream of the dam affected by warming. I then determined the relative role of landscape variables and dam characteristics in driving inter-site differences and the effects of flow on within-site variation in thermal response. Given the importance of temperature to stream ecosystems, these results will help managers prioritize dam removal sites that will maximize thermal benefits to dam removal and promote removals with the intended goal of restoring natural, coldwater thermal regimes.

My third chapter quantifies impacts of small, surface-release dams to DO regimes. The impacts of small dams to DO have been even less studied than those of temperature, although there is again a dichotomy of impact between studies reporting no effect and those showing a large effect on downstream DO. My goals with this chapter were to quantify the impacts to oxygen concentrations and daily oxygen ranges in the impoundments and downstream relative to upstream conditions. I used features of both the landscape and the dams to explain the observed differences in response across sites. As water quality improvements are often cited within dam removal project proposals, understanding which sites are likely to experience negative effects to oxygen regimes can help managers establish more accurate expectations following dam removal.

Table 1.1 – Predictor variable data, data sources, and hypothesized relationship with the number of dams removed within a state. Annual rainfall is a 30-year average (1971-2000), and political affiliation was determined based on voting for the same political party (democratic, abbreviated as "Dem." and republican, abbreviated as "Rep." in table) in at least three of the past four presidential elections.

Data	Source	Hypothesized relationship
Number of dams	National Inventory of Dams 2016	Positive
Dam density (dams/area)	National Inventory of Dams 2016	Positive
Gross domestic product	Bureau of Economic Analysis 2016	Positive
Annual rainfall	National Oceanographic and Atmospheric Administration	Positive
Political affiliation	National Archives and Records Administration	Dem. more than Rep.

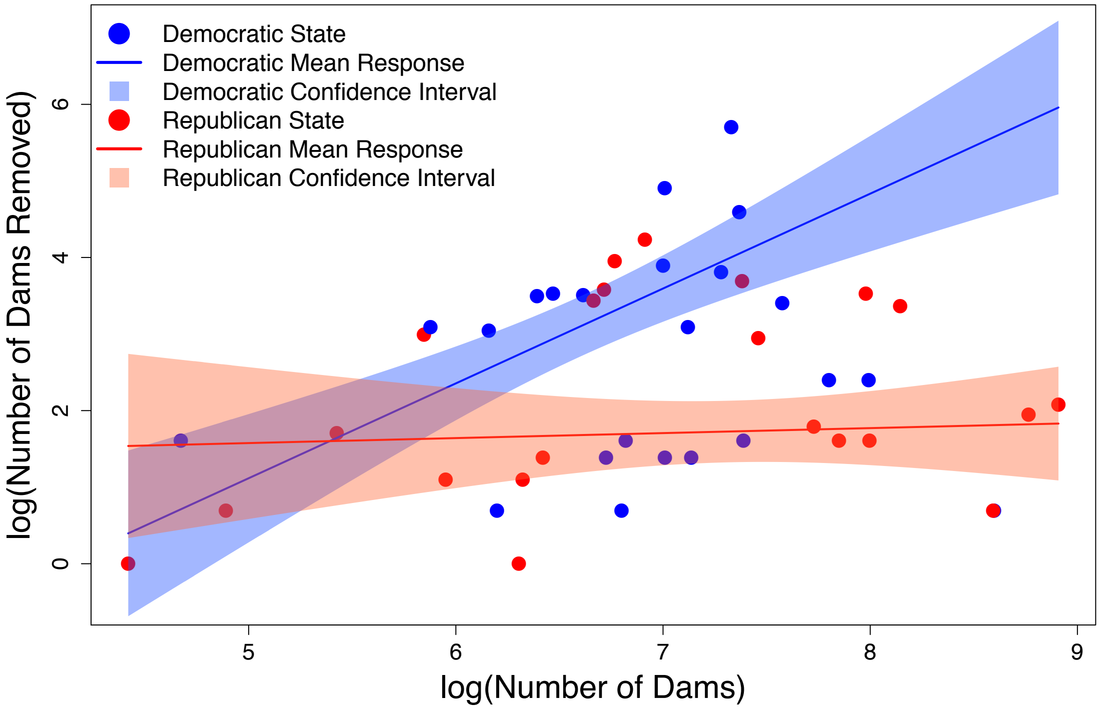

Figure 1.1 – The number of dams removed within a state was best predicted by the number of dams listed within the National Inventory of Dams (2017) for that state and whether the state voted republican or democrat in the last four presidential elections (2004, 2008, 2012, 2016). Each point is a single state (color-coded by political affiliation) and the dark lines are the mean response between the number of dams and number of dams removed for that political party with the affiliated shaded region corresponding to the 95% confidence interval about that mean relationship.

CHAPTER 2

VARIATION MATTERS: INVESTIGATING DRIVERS OF INTER- AND INTRA-SITE DIFFERENCES IN THERMAL RESPONSES TO SMALL DAMS

2.1 Introduction

For the past several centuries, free-flowing streams and rivers of the United States have been altered by an impressive campaign to control and harness their energy through damming. The nation's most comprehensive database of dams, the National Inventory of Dams (NID), managed by the U.S. Army Corps of Engineers, currently lists >90,000 dams in the country (USACE 2016). However, as the NID only contains higher risk structures (dams must meet size requirements of being >25 feet tall with 15 acre-feet of storage or 6 feet tall with 50 acre-feet of storage or be categorized as a high or significant hazard dam), a vast majority of dams are not included; estimates put the actual number of dams in America as high as 2 million (Graf 1993). This estimate considers the number of small dams in the country, which are more prevalent, less documented, and more poorly studied than their larger counterparts (Graf 1999, Poff and Hart 2002, Magilligan et al. 2016), and suggest that the nation's streams may be more altered and more negatively impacted by dams than previously considered.

Dams, regardless of size, have been shown to have copious negative impacts to stream ecosystems (Bednarek 2001). Some impacts are inherent to damming a stream, such as the conversion of a lotic stream reach into a lentic reservoir, blocked upstream and downstream fish passage, and the impoundment trapping of downstream sediment. While these impacts can be alleviated in some cases via structural changes (e.g., building a fish ladder can improve fish passage; Noonan et al. 2012) and operational changes (e.g., high flow releases can pulse trapped sediment to degraded downstream reaches; Melis et al. 2012), such impacts are nearly ubiquitous at all dam sites.

The effects of dams on water quality may be highly variable based on features of the dam, particularly where the dam releases water. Generally, dams release water from one of two depths in the reservoir: the epilimnion (water surface) and the hypolimnion (reservoir bottom). Bottom-release dams are generally tall enough structures for water to stratify within the reservoir, leaving the coldest and most oxygen-poor water in the hypolimnion. Given that the hypolimnion of most reservoirs remains ~4 °C year-round, the controlled releases (e.g., for hydropower, flood storage, etc.) from this layer of the reservoir, result in predictable and consistent changes to downstream water quality. Hypolimnetic dams release unnaturally cold water downstream in the summer when stream temperatures are warm and conversely release unnaturally warm water downstream during the winter (Holden 1979, Ward and Stanford 1979, Armitage 1984). Given the consistent temperature of the released water, these releases often result in decreased diurnal variation in the tailwaters (Lowney 2000).

In contrast to the consistent impacts of bottom-release dams to temperature, surface-release dams have been shown to have highly variable downstream thermal effects. Most small dams are surface-release structures and this variability makes determining the downstream thermal impact of small dams difficult. These dams are thought to generally warm downstream waters as a result of increased solar radiation in the impoundment and the mass of water within the impoundment diluting or disrupting entirely cool subsurface flows (Bednarek 2001). However, this downstream warming effect is not always the case— while some studies have demonstrated downstream warming effects from surface-release dams (Lessard & Hayes 2003, Saila et al. 2005), others have shown little to no impact of these dams on downstream temperature (Bushaw-Newton et al. 2002, Stanley et al. 2002, Smith et al. 2017). The reasons for this variability in downstream thermal responses to small dams have not been investigated, and the studies that have looked at thermal effects of small dams often investigate only a small number of dams and are limited in their spatial extent of stream studied. Those studies with longitudinal sampling downstream of dams provide further insights into the extent of impact and demonstrate that warming can persist for a long distance downstream of a dam, and in many cases may not recover to upstream conditions within the study reach (Fraley 1979, Lessard and Hayes 2003, Bellucci et al. 2011, Dripps and Granger 2013).

Temperature plays a fundamental role in aquatic ecosystems, and dams that alter stream temperature can have an impact on aquatic organisms. Aquatic organisms are ectotherms that are unable to self-regulate body temperature, meaning that biological functions and processes are highly dependent on external stream temperatures. Species vary in their thermal optima, or preferred temperatures for growth and aerobic capacity, as well as their thermal limits, which is operationally the temperature at which half of the population is expected to die, regardless of acclimation (Brett 1952, Brett 1956, Eliason 2011). On an individual level, warmer waters within a species' thermal optima will lead to higher metabolic rates and increased feeding to sustain the associated elevated energy expenditure (Jobling 1997). These increased feeding rates may lead to faster growth rates (Walther et al. 2006, Singer and Gangloff 2011), though this relationship is highly dependent on season, flow rates, age-class, and density (Xu et al. 2010, Letcher et al.

2015). At a population level, thermal optima and thermal limits determine species distributions (Jacobsen et al. 1997, Olden and Naiman 2010), and changes to natural thermal regimes, particularly in the face of a changing climate, may play a large role in biotic impacts and species distributions of the future (Isaak et al. 2017). Temperature can also cue the timing of such life history events as migration, spawning, and egg-hatching (Gahagan et al. 2010, Olden and Naiman 2010), and thus temporal shifts in temperature can have catastrophic potential impacts for populations.

Beyond its impacts to individuals, temperature can also impact stream ecosystem processes. Warm water physically cannot hold as much oxygen as cool water. Low oxygen can be both stressful to aquatic organisms and shift in-stream metabolism from being photosynthesis-dominated to being respiration-dominated, which can result in a system that is more reliant on and therefore retaining greater amounts of outside organic matter (Fisher and Likens 1973, Allan 1995). Increased temperature can also spur harmful algal blooms that can cause oxygen depletion upon die-off (Smith and Schindler 2009, Przytulska et al. 2017). This bottom-up impact, in addition to top-down impacts from thermally-induced predation, carry the potential to drastically change food webs (Perkins et al. 2010, Shurin et al. 2012). These thermal and ecological impacts helped to identify dams as key disruptors of previously well-established longitudinal gradients of processes through river systems (i.e., the River Continuum Concept; Vannote et al. 1980) that lead to fundamental changes in stream ecological understanding (i.e., the Serial Discontinuity Theory; Ward and Stanford 1983).

Given the ecological significance of stream temperature, along with the prevalence of small, surface-release dams in New England, I sought to characterize how small dams impact stream temperatures. Insights into the thermal impacts of small dams have been garnered from studies that considered a limited spatial extent at a small number of dam sites, with the largest number of dam sites in any one study being 10. This study expands on that literature body by using the same collection methods at 30 dam sites, thereby allowing for an assessment of factors influencing responses at a much larger spatial scope than prior work. Specifically, my objectives were to (1) quantify the downstream thermal responses to small dams, (2) investigate the relative effect of landscape variables and dam characteristics as drivers of inter-site variation in thermal response, and (3) examine the effects of flow on the intra-site variability in daily summer downstream temperature impacts. Understanding the factors that drive inter- and intra-site variation in thermal response to small dams can allow managers to predict more accurately how resilient stream reaches are to changing climate and which streams have the greatest potential for improvement following restoration.

2.2 Methods

2.2.1 Study Area

I assessed the impacts of small dams on temperature at 30 surface-release dam sites in Massachusetts, USA (Figure 2.1). The 30 dams in this study could be broken down into four groups: beaver dams (n= 3), winter drawdown (n= 4), unmanaged run-of-river (n= 15) most of which were former mills with no current use, and water supply reservoirs (n= 8) (Table 2.1). These dams were distributed across the state, with terrain ranging from mountainous, high gradient slopes (maximum mean watershed slope of a dam in this study of 16%) in the western part of the state to low gradient coastal plains (minimum mean watershed slope of a dam in this study of 3%) in the east (Massachusetts Bureau of Geographic Information (MassGIS) Digital Elevation Model 1:5,000). As one of the most densely populated states in the country (US Census 2010), Massachusetts has watersheds ranging from highly forested (least impacted site in this study with 95% watershed forest cover), to highly urbanized (most urbanized site had 26% watershed forest cover and 28% impervious cover). Landscape features for all sites are detailed in Table 2.2.

2.2.2 Study Design

I deployed two to six temperature loggers downstream of each dam in this study, with the length of the downstream reach, number of loggers, and spacing between them determined by site-specific characteristics, including access. The first downstream logger was installed at the first suitable location (deep pool with suitable bank structure to secure logger) ~35m downstream of the dam while the most downstream location was upstream of a major confluence, reservoir, or estuary or just upstream and downstream of an inflowing tributary. In addition to the downstream loggers, sites with a single, main tributary flowing into the impoundment (n=18) had an upstream reference logger deployed above the influence of the impoundment, and 15 sites had a logger deployed within the impoundment near the spillway (Figure 2.2).

I used HOBO® Water Temp Pro v2 data loggers (U22-001; Onset Computer Corporation, Bourne, MA). Loggers were deployed in white PVC flow-through housings to both physically protect them and shield them from direct solar radiation (Dunham et al. 2005). A field thermometer was used to ensure that installation locations were representative of each sampling reach. Within free-flowing stream reaches, loggers were deployed in deep pools or runs to ensure submersion throughout summer low flows. Impoundment loggers were deployed at the water surface to capture the temperature of water spilling over these epilimnetic-release dams.

Loggers were deployed year-round beginning in summer 2014 (n= 16), in summer 2015 (n= 11), and summer 2016 (n= 3) and set to record temperature every 15 minutes. As analyses focused on summer temperatures, there were 16 active sites in the summer of 2014, 26 active sites in the summer of 2015, and 20 active sites in the summer of 2016. Logger accuracy was checked annually via an ice bath (Dunham et al. 2005) and loggers were visited biannually (spring and fall) to offload data. At each site visit, a spot check temperature with a National Institute of Standards and Technology (NIST)-certified thermometer ensured accuracy. Temperature data were visually and quantitatively checked using the ContDataQC package in RStudio (RStudio Team 2016) to identify anomalous points and periods where the logger may have come out of water. Anomalous data were flagged and removed from analyses.

2.2.3 Landscape Variables, Dam Features, and Flow Data

I used the United States Geological Survey (USGS) application StreamStats (http://streamstats.usgs.gov) to delineate each watershed with the dam as the outlet. For each watershed, several landscape variables were calculated using StreamStats, including: watershed size, mean watershed slope (USGS National Elevation Dataset 2007), mean watershed elevation (USGS National Elevation Dataset 2007), percent forest cover (MassGIS Land Use 2004), percent impervious cover (National Land Cover Dataset 2011), percent open water (MassGIS Massachusetts Department of Environmental Protection (MassDEP) Wetlands 2009), percent wetlands (MassGIS

MassDEP Wetlands 2009), and the percent underlain by sand and gravel (MassGIS Surficial Geology 2004). Reservoir area was measured in ArcGIS 10.3 (Environmental Systems Research Institute, Redlands, CA) from the MassGIS MassDEP Hydrography 1:25,000 layer. Dam heights were obtained through the NID, technical reports, or by measuring the change in elevation from upstream-downstream of the dam in ArcGIS using MassGIS Digital Elevation Model 1:5,000. Impoundment volumes were obtained through the NID or technical reports, or estimated based on impoundment surface area and maximum depth (assuming a half-sphere shape). I included a binary factor (yes/no) for if the dams had some feature (e.g., secondary spillway, fish ladder, significant hole in the dam) present that would allow a high amount of water to pass, based on visual observations in the field.

To determine the impact of flow events on how temperature changes downstream of a dam, I used discharge data from USGS gages across the state. I chose a suitable gage for each dam site that was either on the same stream as the dam or chose a substitute gage based on similar characteristics (e.g., watershed size, geographic proximity) with the dam site. In both cases, I calculated the ratio between the size of the two watersheds upstream of the gage and upstream of the dam, and used that ratio to adjust the gage discharge data to more accurately reflect the discharge at each dam.

2.2.4 Data Analysis

I calculated daily minimum, mean, and maximum, and monthly mean temperatures for each of the loggers in this study throughout the study period. I used these values to calculate the magnitude of downstream warming, which was the value of a downstream logger minus the upstream logger, such that positive values indicated warmer downstream temperatures than upstream. To avoid potentially high rates of groundwater influx immediately downstream of the dam, I used a logger that was between 40-180 meters downstream of the dam for downstream-upstream comparisons. Autocorrelation tests were used to thin daily data and avoid temporal autocorrelation that otherwise would result in inflated significance values. I used paired t-tests (by date) to determine significant downstream warming magnitudes at each dam site. I also calculated a downstream temperature decay rate as the slope of the linear or logarithmic (whichever had the better fit) line between the mean temperature value of each downstream logger and the logger's distance from the dam. For sites with a surface impoundment logger, I included impoundment temperature in the decay rate regression as well. For these calculations, I assumed the temperature at the impoundment logger equaled the temperature of the water spilling over the dam and listed the distance at 0.001 km downstream of the dam. Because I was interested in cooling rates as a means of recovery from hypothesized warming, I used a one-tailed t-test to determine significance in the mean daily downstream decay rates. Where possible (e.g., when a site had significant warming and a significant negative decay slope), I combined the warming magnitude and decay rate (whichever regression had the better fit – linear or logarithmic), and solved for the distance where the downstream temperature would equal the upstream temperature (e.g., with the assumption that achieving the upstream temperature downstream would be recovering the warming effect) to determine a thermal footprint downstream of the dam. I compared the mean warming magnitudes across all sites by month using a one-way ANOVA. Using thresholds established by Beauchene et al. (2014), I classified upstream and downstream reaches in this study into one of three thermal classes – coldwater

(<18.45 °C), coolwater (18.45–22.30 °C), and warmwater (>22.30 °C). The authors of the paper used July data to create these classifications, while I used August as the majority (n= 10) of the 18 sites classified were deployed mid-July 2015 and would not have had sufficient data to use July temperatures. The patterns of classification were similar across both months.

I used both univariate linear regression models to determine how landscape and dam characteristics affect differences in downstream thermal response (Objective 2). All continuous predictor variables were Z-score standardized to eliminate significance due simply to differences in the scales of units amongst predictor variables within multivariable modelling. I regressed the mean August warming magnitude and mean August downstream linear decay rate as response variables in individual regression models against each of the 11 landscape and dam characteristics separately to directly compare the relative effect of each variable on downstream response metrics. I used both pairwise correlations and variance inflation factors (VIF) to identify collinearity amongst predictor variables, and did not include predictors with correlation coefficients >0.7 and VIF values >3 in further additive models. I then tested a series of separate linear mixed effects models for each of two response variables – mean August warming magnitudes (n= 32) and mean August linear decay rates (n= 50) – against additive combinations of landscape variables and dam characteristics as predictors. Site and year were tested as random effects to focus on inter-site variation in downstream response. I capped the number of predictor variables (not including the random effects) tested for each response variable at three to avoid overfitting the models. Given that a driving force behind understanding the variation in dam impact was prioritizing for dam removal, I did not include the 3 natural beaver dams and only considered the drivers of variation in linear decay rates downstream of the 27 manmade dams. Akaike information criterion corrected for small sample size (AICc; Burnham and Anderson 2002) was used to determine the best supported model. I used a Welch's ANOVA to test for significance between decay rates downstream of sites with and without an auxiliary spillway as the test is not sensitive to highly unequal variances between groups.

For Objective 3, I used daily summer (22 June – 21 September) discharge from USGS gages as the predictor variable in univariate linear regression models with both the daily magnitude of downstream warming and daily downstream decay rate as response variables in separate models for each site. I used R version 3.3.1 (R Core Team 2016) to perform all analyses. Results were considered significant if $p < 0.05$.

2.3 Results

2.3.1 Downstream Thermal Responses to Small Dams

Warming magnitudes downstream of the 18 dams in this study with un upstream logger were highly variable across months. The one-way ANOVA comparing mean downstream warming magnitudes across all dams by month indicated that the greatest downstream warming occurred in June – September (Figure 2.3). There was a moderate amount of warming in May and October (the two months book-ending the high-warming block), and very little changes in downstream temperature from November – April. Given the similarly high magnitudes of warming between June – September and the fact that August had the most complete months of data out of that block (due to timing of initial deployments and needing to pull loggers prior to dam removal), I focused subsequent analyses on August data.

All but one (CGM, a site that was dewatered several years ago for safety reasons) of the 15 sites with an impoundment experienced the warmest mean daily August temperatures within the impoundment. The warm impoundment waters translated to warmer downstream waters relative to upstream temperatures at the majority (61%) of sites, with mean daily August warming values ranging from 0.20 – 5.25 °C (average= 1.51 °C ; Figure 2.4). Five sites had no significant differences between upstream and downstream temperatures, and 2 sites experienced cooler temperatures downstream of the dam than upstream (Figure 2.4). Using the averaged upstream August temperatures from each site across its deployment period as the would-be condition if the dam were not present, there were two coldwater sites (<18.45 °C), seven coolwater sites (18.45 – 22.30 °C), and nine warmwater sites (>22.30 °C) in this study (Table 2.3). Downstream reaches were either coolwater habitat (n= 4) or warmwater (n= 14), there were no sites with coldwater habitat downstream of the dam. Both upstream coldwater sites were warmed to downstream coolwater, and five of the upstream coolwater sites were warmed to downstream warmwater reaches, while two upstream coolwater and nine upstream warmwater sites experienced no difference between upstream and downstream reaches. Of 32 upstream classifications (non-averaged data), 12 were shifted to a warmer downstream thermal class by the dam (Table 2.3).

Twenty-seven sites had multiple loggers that remained submerged throughout the summer from which a decay rate could be calculated. All but four of those 27 sites (85%) had temperatures that significantly decreased with increasing distance downstream of the dam in at least 1 year of the study (Table 2.4). Mean August monthly decay rates for sites with linear decreases ranged from -0.46 °C/km downstream of WBH in 2014 to -9.50 °C/km downstream of MOO in 2016 (average= -2.08 °C/km; Table 2.4). Across the multiple years of study of the 27 sites with thermal decays downstream of the dam (totaling 48 mean August decay rates), 30 of those decay rates were best fit by a linear regression (Figure 2.5a), whereby temperature cooled at a constant rate with distance, and 18 were best fit by a logarithmic regression (Figure 2.5b), whereby temperature change was most pronounced immediately downstream of the dam and flattened out (i.e., decreased at a slower rate) further downstream. There was no relationship between temperature and distance downstream of one site (MAR; Figure 2.5c), and three sites had temperatures increase with increasing distance downstream of the dam (e.g., PIC; Figure 2.5d).

Fifteen sites had both an upstream logger and multiple downstream loggers (to determine a decay rate), and only 7 of those sites had both significant warming caused by the dam and a significant cooling rate downstream of the dam that together could determine the downstream extent of warming (i.e., the thermal footprint) (Table 2.5). One of these sites had an estimated footprint of >1,400 kilometers, ensuring that recovery at the observed decay rate would not be met at that site. Three dam sites had a footprint <1 kilometer, while the remaining three had mean August footprints of 1.3, 2.8, and 4.8 kilometers. Not including TUR (with the >1,400 km footprint), the average downstream footprint at the remaining six sites was 1.7 kilometers.

2.3.2 Drivers of Inter-Site Variation

The small dams in this study were widely distributed across the state of Massachusetts (Figure 1) and varied widely in dam features and physical settings, allowing for investigations into the drivers of differences in downstream response that has not been previously studied. Dams were an average of 5.3 m high (range: 0.4 – 15.0 m) and had impoundments with an average surface area of 32.4 ha (range: 0.1 – 261.9 ha) and average volume of 1,426,811 m^3 (range: 200 – 13,568,280 m^3) (Table 2.1). The average watershed was 58.4 km^2 (range: 0.5 – 388.5 km^2) in size with 66.9% (range: 23.5 – 95.4%) forest cover with an average mean basin elevation across the sites of 221.1 m (range: 23.8 – 448.1 m) above sea level (Table 2.2).

Several of the individual landscape and dam variables significantly predicted the magnitude of downstream warming. Forest cover and dam height being positively related to the magnitude of warming, with more warming in more forested watersheds and downstream of tall dams (Table 2.6). Watershed size, impervious cover and the percentage of the watershed underlain with sand and gravel had negative relationships with warming (Table 2.6). Several predictors (mean watershed elevation, impervious cover, percent sand and gravel, mean watershed slope, and impoundment surface area) had VIFs >3 and correlation coefficients >0.7 and thus were not included in the additive models. The best supported model (moderate support; marginal R^2= 0.51) for explaining downstream warming magnitude included watershed size (negative) and impoundment volume (positive) with a random effect for site (Figure 2.6). As such, this model suggests that the largest downstream warming magnitudes occurred downstream of dams impounding large amounts of water in small watersheds (Figure 2.6).

None of the individual landscape feature of dam characteristic variables could significantly predict downstream decay rates (Table 2.6). However, the presence of an auxiliary spillway did have a significant effect on decay rates, and sites with an auxiliary spillway experienced slower downstream decay rates than those without a spillway, Welch's F(1,45.787) = 15.833, $p< 0.001$). Mixed effects modeling determined a model consisting of watershed forest cover (negative) and dam height (positive) with a random effect for site best explained the variation in downstream decay rates (Figure 2.7), albeit only modestly with a marginal R^2 of 0.38. This model suggests that the fastest cooling rates (most negative decay rates; <0 °C/km) occurred downstream of short dams without an auxiliary spillway in highly forested watersheds (Figure 2.7).

2.3.3 Influence of Flow on Daily Summer Intra-Site Variation

Downstream response metrics ranged not only across sites but also within, as daily response metrics at each site exhibited a range of values throughout the summer (22 June – 21 September). For both the downstream warming magnitude and the decay linear decay rate, there were sites that experienced little change in the daily response metric throughout the summer, and others that experienced a wide range in daily response metrics. The spread in downstream warming magnitudes ranged from 1.26 °C between the largest and smallest amount of warming observed downstream of a dam (LRM) to 6.61 °C difference between warming magnitudes throughout the summer (CRA). The stability of the downstream warming magnitude was related moderately strongly to the mean upstream temperature ($r^2= 0.63$), with warmer sites experienced less variation throughout the summer than cooler sites, that experienced much larger changes in daily warming magnitudes throughout the summer. The range in daily downstream linear decay rates was also highly variable across throughout the summer at a single site was also highly variable across sites. At the site with the most stable decay rates across the summer (TUR), decay rates only changed 1.04 °C/km, while at the most variable site (UPN) decay rates varied by 26.74 °C/km, as the site often experienced steep (e.g., >-10 °C/km) cooling rates, but occasionally experienced patterns in which temperature increased downstream of the dam.

Mean daily discharge had a significant effect on the differences in daily warming magnitudes at the majority (10 of 18) of the sites in this study (Table 2.7). Eight of those were sites that experienced mean August warming, and on a daily scale, experienced greater magnitudes of warming during periods of low flow (see Figure 2.8a). Eight sites did not experience a relationship between flows and downstream warming magnitudes (see Figure 2.8b). Two sites (BOS and CGM) were the only two sites in the study with significant downstream cooling and experienced the opposite (positive) relationship between flows and warming such that they experienced greater cooling during low flows (see Figure 2.8c). All sites that were affected by flow experienced greater thermal similarity between upstream and downstream reaches during high flows.

Seventeen (63%) of the 27 sites with multiple loggers submerged downstream (Table 2.4) had downstream linear decay rates that were significantly affected by mean daily discharge rates (Table 2.7). Six sites had daily decay rates that were negatively related to daily discharge, meaning that they experienced faster cooling rates (steeper decay rates) during high flow events (see Figure 2.8d). Two of the three sites that experienced increasing temperatures and the one site that had no relationship with distance downstream of a dam experienced this negative relationship with flow. The linear decay rates of 10 sites were not affected by discharge (see Figure 2.8e). The most common relationship between flow and decay rates was a positive relationship observed at 11 sites, whereby there was less of a decay (slope ≈ 0 °C/km) during high flow events (see Figure 2.8f).

2.4 Discussion

2.4.1 Variation in Downstream Response

I observed a wide variation in downstream warming to small, surface-release dams in this study, with mean daily downstream August temperature differences relative to upstream ranging from -1.2 – 5.3 °C. Results from this study paralleled those reported across the collective literature (Figure 2.9), in which downstream temperatures ranged from a minimum of -1.0 °C (i.e., downstream cooling; Lessard and Hayes 2003) to a maximum of +6.6 °C (i.e., downstream warming; Maxted et al. 2005) different than upstream temperatures. The variation in downstream responses observed across the literature called into question the accuracy of the blanket statement that surface-release dams warm downstream waters reported in review papers (e.g., Bednarek 2001). However, the small number of dam sites per study previously published (average≈ 3 dams/study) made it difficult to be sure of what the effect of surface-release dams on downstream temperature was. This was especially problematic when those results that reported no change in downstream temperatures often come from a study of a single dam (Bushaw-Newton et al. 2002, Stanley et al. 2002), and could easily be viewed as unique or outlying values. A major benefit from this study is the large increase (~6x over the prior average) in the number of dam sites (n= 18) into how dams alter downstream temperatures relative to upstream temperatures. The results from these 18 sites indicate that reported results of no effect from studies of a single dam are not outlying values, but instead are important in understanding there is a wide range of downstream thermal responses to small, surface-release dams and that the generalization of warming by surface-release dams reported in review papers is an overly simplistic and often inaccurate generalization.

A second problem with the blanket statement of warming downstream of surface-release dams is that it discounts the wide variation in the magnitude of warming below these dams and what may be driving these differences. The warming magnitudes downstream of the 11 sites within this study that experienced downstream warming were highly variable, and ranged from 0.20 – 5.25 °C (mean= 2.17 °C) warmer downstream than upstream temperatures. This variation in downstream response seems largely to be driven by a dam's landscape position, with dams on smaller, forested headwater streams likely to have a larger downstream warming effect than dams on more urbanized rivers in larger watersheds. These results support the hypothesis of Jones et al. (2010) that, all else equal, headwater stream dams would have a larger warming effect than dams on larger rivers. Stream temperature is largely a function of the amount of energy in a volume of water (Poole and Berman 2001, Caissie 2006), and small stream reaches immediately downstream of a dam are likely more susceptible to warming from a single input (e.g., an impoundment) introducing relatively large amounts of warm water. This warming effect may be driven by the larger relative difference in canopy cover between the upstream reference reach and the impoundment in small rivers, whereas upstream canopy cover in larger rivers is lower resulting in less of an impact of the impoundment on canopy. There is a large body of research investigating the impacts of canopy cover loss as a result of logging practices (Johnson and Jones 2000, Quinn and Wright-Stow 2008, Janisch et al. 2012) that have demonstrated significant warming (up to 3.8 °C; Quinn and Wright-Stow 2008) as a result of a clear cut. Beyond canopy cover loss, headwater streams may be more susceptible to warming because they have a greater potential for warming, based on the idea of thermal equilibrium. Sites on larger rivers (i.e., those with larger watersheds, >100 km^2 such as BAL and BOS on the Shawsheen and Ipswich Rivers, respectively) in this study had temperatures much higher than the smaller streams, and likely existed at temperatures closer to the equilibrium temperature for the region.

2.4.2 Biotic Implications

The increased temperatures caused by dams within this study can have biological impacts for downstream ecosystems. Most directly, warming downstream waters can change species distributions below dams, and shift temperatures out of the thermal conditions many aquatic ectotherms have evolved in (Allan 1995). Warming waters have the potential to most negatively impact coldwater species, and most positively impact warmwater species. Given that several studies have observed decreases in the abundance of coldwater species co-occurring with increases in warmwater generalist in downstream waters warmed by a surface-release dam (Lessard and Hayes 2003, Hayes et al. 2006; Bellucci et al. 2011), it is reasonable to expect that these same community effects could occur following warming downstream of the dams in this study. While fish species data were not collected in conjunction with the temperature monitoring in this study, classifying the temperature of upstream and downstream reaches for the dams in this study can approximate how species might be impacted by these thermal changes. From the cold-, cool-, and warmwater classifications developed by Beauchene et al. (2014), seven sites experienced a shift from a cooler thermal class upstream to a warmer class downstream of the dam (i.e., coldwater to coolwater or coolwater to warmwater) that would suggest a change in fish community composition. The 11 sites (two coolwater, nine warmwater) that did not experience thermal class shift from upstream to downstream of the dam likely had minimal effects on fish community composition.

2.4.3 Spatial Extent of Downstream Impact

Streams are not static environments, and downstream reaches that experienced warming as a result of a dam are expected to cool with longitudinal distance downstream associated with groundwater inputs or sufficient shading. The patterns and rates of temperature change downstream of the dams in this study were highly variable across sites. Most of the streams in this study had temperatures cool following a more linear pattern, experiencing a consistent rate of temperature decline regardless of the distance downstream of the dam. A number of sites within this study also, however, had temperatures cool following a more logarithmic decay pattern, whereby the greatest temperature change occurred immediately below the dam, and there was less of a decay (i.e., slower cooling) with distance further downstream of the dam. Both of these patterns were observed in the small body of literature that could offer insights into the patterns of temperature cooling downstream of small dams (Bellucci et al. 2011, Dripps and Granger 2013). It should be noted that neither of these studies actually fit decay curves to the temperatures downstream of the dams in their studies, and I glean insights from the apparent patterns displayed in figures (see Figure 10 in Bellucci et al. 2011 and Figure 5 in Dripps and Granger 2013) within these papers. Of four dams between the two studies, one seems most likely to be fit by a linear decay, two certainly by a logarithmic curve as they experience a sharp decrease immediately below the dam followed by a more linear and slow decay pattern, and the fourth may be best fit by either type of curve. Neither of these two studies calculated the rate of thermal decay, and while Maxted et al. (2005) did not report on the patterns of decay, the authors of that paper stated that temperature cooled at roughly -10 °C/km downstream of several small dams. The average cooling rate downstream of the dams within this study was -2.08 °C/km, but encompassed a wide range from very little cooling (-0.46 °C/km) to very rapid cooling (-9.50 °C/km) similar to the rates observed by Maxted et al. (2005).

The pattern and rate of cooling can determine how far downstream a dam's warming effect will persist (i.e., the dam's downstream thermal footprint). In this study, one site had a footprint estimated to be >1,400 km, likely due to a slow logarithmic decay rate (i.e., quick initial cooling followed by a slow cooling thereafter) observed downstream of the dam. Discounting this anomalously large distance, the average downstream thermal footprint across the remaining six sites at which this calculation was possible was 1.7 km. I am not aware of any other study that has directly calculated the thermal footprint downstream of a small dam, though several studies report distances at which elevated temperatures downstream of a small dam were or were not recovered. These "no-recovery" distances range from 0.5 km (Dripps and Granger 2013) to ~3 km (Lessard and Hayes 2003, Bellucci et al. 2011), with one study reporting elevated temperatures >50 km downstream of a surface-release dam (Fraley et al. 1979). Maxted et al. (2005) reported that elevated temperatures below the dams in their study persisted for several hundred meters given the very rapid downstream cooling rates of -10 °C/km.

As most of the cooling rates in this study were not that fast, it seems sensible that the average footprint in this study would extend a bit further downstream. While it did not consider temperature impacts, a study of geomorphic footprints by Fencl et al. (2015) measured an average downstream footprint of 1.2 km below six low head dams adding support to the average footprint distances obtained via this study.

2.4.4 Monitoring Temperature Impacts

Understanding the impacts of dams to stream temperature and the drivers of variation in impact is helpful in determining dam sites likely imparting the largest thermal impacts on a stream. However, monitoring is necessary to determine the dam-induced impacts at a site, help to establish realistic expectations for recovery following restoration activities (e.g., dam removal), and determine if those goals are being met with post-restoration monitoring. Currently, <10% of dam removals monitored scientifically, with water quality (e.g., temperature) receiving some of the least amount of attention (Bellmore et al. 2017), leaving large gaps in our collective understanding regarding stream thermal impacts and predicted responses to dams and dam removal. Dams exhibited the largest downstream impacts during warm, summer months (June – September), and the majority of dams exhibited the largest downstream impact within those warm months during periods of lowest flow. Given these relationships, temperature monitoring efforts in New England should be focused during warm summer months and during periods of low flow to identify the 'worst case' effect of a dam. This study focused on August data and did not see an effect of year and suggests that a single month of summer monitoring, in most cases, may be sufficient to quantify the impacts of a dam. The Stream Barrier Removal Monitoring Guide calls for monitoring temperature at a minimum of one location upstream of the impoundment (reference), at one location within the impoundment, and at one location downstream of the dam (Collins et al. 2007). To better understand not only the single downstream impact, but also how far downstream thermal impacts persist for, temperature monitoring should utilize cheap high-resolution continuous temperature loggers (see U.S. EPA 2014 for discussion into commercially available loggers) at a slightly expanded spatial extent downstream of the dam. In addition to the upstream and impoundment loggers, downstream temperature monitoring at 3-4 locations downstream of the dam until a thermal barrier (e.g., tributary, estuary, impoundment) is reached will allow managers to understand more completely the extent of thermal alterations.

2.4.5 Prioritizing Restoration Efforts

Identifying factors that lead to greater downstream thermal effects below small dams can help to better prioritize restoration efforts. Stream restoration is a huge industry, with the United States spending approximately $1 billion per year on improving stream and river habitat (Bernhardt et al. 2005). With ever-shrinking budgets for maintaining and restoring aquatic habitat integrity, models and methods for prioritizing restoration efforts are critical for best using limited financial resources (Branco et al. 2014, Hoenke et al. 2014). These thermal impacts may give higher priority to dam removal as a restoration tool over other forms of aquatic connectivity restoration (e.g., poorly designed road crossings; Nislow et al. 2011), as dam impacts on temperature likely extend both upstream (as a barrier to aquatic organisms) and downstream (warmed downstream temperatures), and present a larger impact to the ecosystem. Understanding broadly that tall dams in smaller, forested streams are having a more negative impact on stream temperature than dams in larger order rivers could direct efforts and funding to remove these dams. I expect that sites experiencing the most negative water quality impacts from the dam will experience the greatest benefit from dam removal.

While many sites in small, forested headwater watersheds experienced the largest impacts, their highly warmed downstream temperatures may still have been suitable for coldwater species. Practitioners aiming to restore coldwater habitat may be best suited to target dams where upstream conditions are suitable for coldwater species, but downstream temperatures are warmed above thermal limits for coldwater species. Dams that are shifting stream temperatures from a coldwater thermal class and into a warmwater class could be identified as sources of thermal pollution. As such, the dam could be listed as a water quality impairment under section 303(d) of the Clean Water Act. Per requirements, a TMDL (total maximum daily load) would need to be developed, and if the dam was listed as the source of impairment, dam removal could be a legally mandated option. While not used often, as it can be expensive and time-consuming, one of the largest proposed dam removal projects in the world (removal of four Klamath River dams in Oregon and California) is using thermal impairment from the surface-release dams as a primary driver for dam removal. And as many state-designated water classes are tied to coldwater fish habitat, listing a dam as a source of thermal impairment may work to also optimize coldwater habitat for fish restoration. One of the sites in this study, Upper Roberts Meadow Dam, shifted the temperature regime of Roberts Meadow Brook from a state-listed Class A (mean of the 7-day max <20 °C) upstream to a class B (mean of the 7-day max >20 °C) downstream; this is a great example of a site that is expected to have an increase in coldwater habitat following dam removal. Alternatively, understanding the drivers of impact may allow practitioners to state more realistic and explicit project goals and understand that some projects. For example, removal of dams on large lowland rivers that will open up miles of anadromous fish habitat may not have any significant water quality improvements.

The majority of sites in this study experienced a homogenizing effect of high flows that worked to minimize the reduce the warming magnitudes to downstream thermal regimes seen during periods of low flow. During low flow periods, an impoundment is likely receiving the greatest amount of solar insolation and experiencing the greatest warming (Fuller and Peckarsky 2011). At the same time, during low flow there is less water in the downstream reach with a subsequent decreased ability to buffer against thermal inputs from an impoundment (Poole and Berman 2001, Caissie 2006). There are several avenues via which this flow-temperature relationship may be useful to managers. The first would be to target periods of low flow for pre-removal monitoring to understand the worse-case temperature effect of the dam in question. Second, this relationship between low flows and increased downstream temperature impact provides additional support to existing restoration prioritization models (e.g., Massachusetts Division of Ecological Restoration’s Restoration Potential Model) that prioritize restoration of dam sites with the greatest reductions in summer stream flow. High temperatures and low flows have been shown to have negative effects in the form of reduced survival of a coldwater adapted salmonid (Letcher et al. 2015), and targeting sites systematically impacted by low flows may provide large biological benefits in addition to improvements in water quality. Differences in a stream’s resilience against or susceptibility to low flow conditions may be a valuable factor when deciding where restoration efforts would have a greater benefit two similar dams in similar watersheds. If dam removal is not possible, I would echo the recommendations of Olden and Naiman (2010) to increase spill rates from dams that are having the largest and most negative effects to downstream thermal regimes as a means of potentially mitigating downstream dam-induced thermal effects.

2.4.6 Conclusion

With a marked increase in the number of dam sites over prior work, this study provides insights into the variation in the thermal impacts of small dams. Most, but not all, of the sites in this study significantly warmed downstream temperatures, and though temperatures cooled with distance downstream of the dam, elevated temperatures persisted on average for over a kilometer below the dam. Warmer waters carry ecosystem impacts that range from the way in which a stream processes nutrients to changes in the community composition of resident aquatic organisms. These impacts underscore the fact that small dams can negatively impact streams and the native flora and fauna that inhabit them beyond the simple fragmenting effect of an instream barrier. Warming magnitudes were most pronounced on streams in small watersheds downstream of dams with large reservoirs. Identifying sites across the landscape that may be most susceptible to the negative thermal effects of small dams can help managers prioritize restoration (e.g., dam removal) efforts and establish more realistic goals following such restoration activities.

Table 2.1 – Features of the dams and associated impoundments. Impoundment type abbreviations are: ‘B’ – beaver dam, ‘D’ – winter drawdown, ‘ROR’ – run-of-river, and ‘WS’ – water supply. Volumes for beaver impoundments are listed as ‘ND’ (no data) as they did not have volume data in a database (e.g., the NID), nor I did not have mean depths to calculate impoundment volume.

			Impoundment			
Name	Site Code	Dam Height (m)	Surface Area (ha)	Volume (m^3)	Type	Auxiliary Spillway
Amethyst - Hawley	AHA	5.2	2.6	123348	WS	No
Amethyst - Meetinghouse Lower	AML	5.0	0.5	10826	WS	No
Amethyst - Meetinghouse Upper	AMU	12.5	2.6	82643	WS	No
Balmoral	BAL	2.1	2.3	22402	ROR	No
Barstow's Pond	BAR	2.6	4.1	259030	ROR	No
Bostik / South Middleton	BOS	3.1	7.5	77709	ROR	No
Ballardvale	BVL	4.3	31.2	357709	ROR	No
Cotton Gin Mill	CGM	1.5	0.1	200	ROR	No
Cranberry Pond	CRA	0.4	11.3	137860	ROR	No
EB Ware R. - Bickford	EBB	15.0	67.4	4164228	WS	No
Hunter's Pond	HUN	3.4	0.5	13568	ROR	No
Ipswich Mills	IPS	3.2	13.1	356428	ROR	Yes
Larrywaug / Stockbridge Bowl	LAR	5.8	155.0	13568280	D	No
Lower Roberts Meadow	LRM	7.0	1.1	47612	ROR	No
Marland Place	MAR	3.8	2.1	20352	ROR	No
Moose Meadow	MOO	12.5	16.3	863436	ROR	No
Middle Roberts Meadow	MRM	12.8	8.9	801762	WS	No
Munn Brook	MUN	4.4	0.5	37004	WS	No
Old Mill	OLD	4.1	3.1	59207	ROR	Yes
Peck's Pond / Onota	PEC	5.5	261.9	8387664	D	No
Piccadilly Brook	PIC	4.6	26.2	1103964	WS	No
Prescott Road 17	PRD	2.0	16.3	NC	B	No
Roaring Brook	ROA	9.1	0.7	39594	WS	No
Tel-Electric	TEL	6.1	4.4	119481	ROR	Yes
(Millie) Turner	TUR	3.1	6.9	80176	ROR	Yes
Underhill Brook	UND	1.2	2.2	NC	B	No
Underhill Brook Tributary	UNT	0.6	0.5	NC	B	No
Upper Naukeag	UPN	2.4	125.0	1541850	D	No
Upper Roberts Meadow	URM	10.7	1.7	80176	ROR	No
WB Housatonic R. / Pontoosuc	WBH	5.8	196.2	6167400	D	No
Min.	NA	0.4	0.1	200	NA	NA
Mean	NA	5.3	32.4	1426811	NA	NA
Max.	NA	15.0	261.9	13568280	NA	NA

Table 2.2 – Landscape characteristics of the watersheds draining to dams. See Table 2.1 for site abbreviations.

	Watershed							
Site	Area (km^2)	% Forest	% Impervious	% Open Water	% Wetland	Mean Elevation (m)	Slope (%)	Sand & Gravel (%)
AHA	3.9	95.4	0.3	0.7	1.7	268.2	10.8	9.3
AML	16.1	92.0	0.3	0.4	5.6	295.1	8.1	14.8
AMU	10.5	91.7	0.3	0.3	7.7	320.0	7.0	14.4
BAL	188.8	25.3	27.6	1.0	12.8	44.5	4.2	52.6
BAR	19.4	66.6	11.9	0.8	14.7	23.8	3.0	44.5
BOS	113.4	31.7	20.6	1.7	20.3	32.9	4.1	52.7
BVL	170.2	25.5	27.7	0.9	13.4	44.2	4.1	53.1
CGM	55.4	41.2	14.9	1.2	19.6	28.5	2.7	32.5
CRA	6.3	80.9	0.8	1.8	2.7	179.2	16.1	22.7
EBB	8.5	85.0	0.3	8.1	6.5	368.8	9.4	0.0
HUN	29.5	72.5	3.2	2.7	21.8	29.9	4.8	13.6
IPS	388.5	49.5	12.7	2.5	19.7	30.2	5.1	46.6
LAR	30.3	51.5	1.3	5.9	9.6	344.4	12.0	0.1
LRM	27.7	84.9	0.2	0.5	5.2	258.5	12.3	14.3
MAR	183.9	25.5	27.4	1.0	13.0	44.5	4.2	53.1
MOO	6.6	78.8	0.4	3.1	8.6	332.2	9.0	6.4
MRM	27.7	85.2	0.2	0.4	5.2	259.1	12.3	14.4
MUN	14.3	92.9	0.2	2.2	0.9	293.8	14.5	7.4
OLD	65.5	46.3	21.3	2.8	11.0	95.1	5.4	36.4
PEC	27.2	64.3	1.5	10.1	3.2	435.9	13.7	12.1
PIC	3.1	23.5	7.6	8.8	5.3	156.1	8.1	29.0
PRD	6.3	79.2	0.2	6.9	3.3	280.4	9.4	0.6
ROA	13.5	91.9	0.2	0.7	3.0	247.2	15.5	2.5
TEL	93.5	59.4	4.5	5.4	4.3	432.8	13.3	12.3
TUR	155.1	78.7	1.5	1.2	6.8	136.6	8.0	21.8
UND	1.7	84.2	0.3	0.0	9.1	277.0	10.7	17.5
UNT	0.5	88.8	0.1	0.0	4.1	285.9	8.7	0.0
UPN	5.1	62.3	0.8	24.9	4.3	359.7	5.4	0.0
URM	22.8	87.3	0.2	0.1	4.9	281.0	12.7	13.3
WBH	56.2	65.0	2.4	4.0	4.5	448.1	14.4	11.0
Min.	0.5	23.5	0.1	0.0	0.9	23.8	2.7	0.0
Mean	58.4	66.9	6.4	3.3	8.4	221.1	9.0	20.3
Max.	388.5	95.4	27.7	24.9	21.8	448.1	16.1	53.1

Table 2.3 – Classification of stream temperatures upstream and downstream of the dams in this study, based on thermal classifications developed by Beauchene et al. (2014). Blue shaded cells represent coldwater, green shading represents coolwater reaches, and red shading represents warmwater. See Table 2.1 for site abbreviations.

	2014		2015		2016		Average	
Site	US	DS	US	DS	US	DS	US	DS
AML	18.39	19.73	19.24	20.89	NA	NA	18.82	20.31
AMU	16.27	21.02	17.47	23.23	NA	NA	16.87	22.13
BAL	NA	NA	23.65	23.67	24.12	23.90	23.89	23.79
BAR	NA	NA	21.79	23.02	22.71	23.87	22.25	23.45
BOS	NA	NA	23.31	23.32	24.82	21.87	24.07	22.59
BVL	NA	NA	NA	NA	24.54	25.08	24.54	25.08
CGM	NA	NA	24.43	23.96	25.03	23.84	24.73	23.90
CRA	17.96	22.24	19.53	24.41	19.83	22.71	19.11	23.12
HUN	NA	NA	21.87	22.64	20.89	24.07	21.38	23.36
IPS	NA	NA	23.69	NA	24.24	24.75	23.96	24.75
LRM	NA	NA	NA	NA	25.75	25.90	25.75	25.90
MAR	NA	NA	23.36	23.63	24.11	24.24	23.74	23.93
MRM	NA	NA	NA	NA	20.72	25.75	20.72	25.75
OLD	NA	NA	23.91	23.83	24.77	24.50	24.34	24.16
ROA	20.24	19.99	21.01	21.60	NA	NA	20.63	20.80
TEL	NA	NA	22.84	22.92	23.51	23.83	23.18	23.38
TUR	NA	NA	21.90	22.81	NA	NA	21.90	22.81
URM	NA	NA	17.95	20.24	18.49	21.70	18.22	20.97

Table 2.4 – Mean August downstream decay rates (°C/km) below each of the dams in this study, with the estimate from both linear and logarithmic decay curves listed. The best fit curve for each site in each year is underlined, with significant ($p< 0.05$) best-fit decay rates in bold. See Table 2.1 for site abbreviations.

	2014		2015		2016	
Site	Linear	Log	Linear	Log	Linear	Log
AHA	**-2.95**	-0.93	-3.28	**-1.07**	-	-
AML	**-4.56**	-0.73	**-4.07**	-0.72	-	-
AMU	**-2.99**	-0.89	**-5.20**	-2.42	-	-
BAL	-	-	-0.43	**-0.04**	-0.55	**-0.07**
BAR	-	-	**-2.81**	-0.16	**-4.35**	-0.27
BOS	-	-	**-3.17**	-0.50	-2.59	**-0.87**
BVL	-	-	-	-	-1.93	**-0.15**
CGM	-	-	**-1.79**	-0.29	-1.67	-0.41
CRA	-3.73	**-1.05**	**-6.18**	-0.68	**-9.35**	-1.19
EBB	0.72	0.43	5.25	2.09	-	-
LAR	**-0.51**	-0.14	-0.45	**-0.17**	-	-
LRM	-	-	-	-	**-3.29**	-0.59
MAR	-	-	-0.37	-0.04	-0.55	-0.03
MOO	**-3.67**	-0.31	-3.66	**-0.36**	-9.50	**-0.84**
MUN	-7.69	**-1.16**	**-9.32**	-1.19	-	-
OLD	-	-	-0.89	**-0.12**	-0.76	**-0.10**
PEC	-0.23	-0.09	-1.33	**-0.35**	-1.17	**-0.36**
PIC	2.21	0.61	2.89	0.79	-	-
PRD	**-2.33**	-0.38	**-1.39**	-0.23	-	-
ROA	**-1.25**	-0.59	**-1.93**	-0.89	-	-
TEL	-	-	-0.97	-0.26	-0.47	**-0.09**
TUR	-	-	-0.23	**-0.08**	-	-
UND	**-1.01**	-0.29	-2.22	-0.94	0.15	0.27
UNT	1.61	0.47	3.92	1.94	7.31	2.12
UPN	**-6.84**	-0.82	-7.72	-0.59	**-9.43**	-0.74
URM	-	-	**-0.43**	-0.10	**-0.76**	-0.18
WBH	**-0.46**	-0.24	**-0.83**	-0.11	-0.69	**-0.40**

Table 2.5 – Predicted thermal footprint (distance to recovery of upstream temperatures based on best fitting downstream decay curve) for seven sites with both warming magnitudes and cooling temperatures downstream of the dam. See Table 2.1 for site abbreviations.

Site	Footprint (km)
TUR	1417.4
URM	4.8
CRA	2.8
AMU	1.3
BVL	0.6
BAR	0.4
AML	0.3

Table 2.6 – Results of univariate models between each of the predictor and response variables tested in this study. Predictor variables were scaled to report relative and comparable effects on the response variables across all parameters. Bold font indicates significant differences at P < 0.05, and '†' signifies predictor variables that were log-transformed for analyses.

	Warming			Decay		
Variable	Estimate	r^2	p	Estimate	r^2	p
Dam Height	**0.96**	**0.19**	**0.041**	0.74	0.05	0.257
Surface Area†	0.25	0.00	0.765	0.71	0.05	0.271
Volume†	1.27	0.07	0.230	0.62	0.05	0.256
Surface Area:Watershed Area†	**2.44**	**0.46**	**<0.001**	0.05	0.00	0.94
Watershed Forest	**1.08**	**0.34**	**0.002**	-1.06	0.11	0.052
Watershed Impervious†	**-1.00**	**0.33**	**0.002**	0.91	0.09	0.105
Watershed Area†	**-1.32**	**0.40**	**<0.001**	0.92	0.08	0.120
Watershed Slope	0.72	0.15	0.094	-0.23	0.03	0.724
Watershed Elevation	0.84	0.13	0.103	-0.24	0.00	0.725
Watershed Open Water†	-1.13	0.16	0.066	0.23	0.01	0.724
Watershed Wetland	-0.69	0.14	0.091	0.64	0.01	0.383
Sand & Gravel	**-0.89**	**0.20**	**0.035**	0.50	0.03	0.406

Table 2.7 – Effects of flow on downstream warming magnitudes and downstream decay rates, with the coefficient of the relationship between the response variables and log-transformed discharge listed ("estimate"). Response direction refers to the directionality of the response variable (warming, linear decay) that was regressed against flow. 'NS' is no significant relationship in the response direction. Significant regression relationships are in bold. See Table 2.1 for site abbreviations.

	Warming				Linear Decay			
	Response Direction	~ log(Discharge)			Response Direction	~ log(Discharge)		
Site		Estimate	r^2	p		Estimate	r^2	p
AHA					-	0.29	0.01	0.196
AML	+	**-0.59**	**0.09**	**0.014**	-	-0.11	0.00	0.839
AMU	+	**-1.02**	**0.19**	**<0.001**	-	**1.90**	**0.39**	**<0.001**
BAL	NS	-0.05	0.00	0.488	-	0.07	0.01	0.163
BAR	+	0.07	0.00	0.338	-	**0.60**	**0.36**	**<0.001**
BOS	-	**0.54**	**0.53**	**<0.001**	-	**-0.46**	**0.23**	**<0.001**
BVL	+	0.29	0.02	0.228	-	**1.60**	**0.38**	**<0.001**
CGM	-	**0.48**	**0.53**	**<0.001**	-	0.19	0.04	0.080
CRA	+	-0.14	0.01	0.183	-	**2.28**	**0.40**	**<0.001**
EBB					+	**-2.11**	**0.49**	**<0.001**
HUN	+	**-0.91**	**0.44**	**<0.001**				
IPS	+	**-0.12**	**0.09**	**0.043**				
LAR					-	**-1.49**	**0.36**	**<0.001**
LRM	NS	0.08	0.00	0.369	-	**1.36**	**0.20**	**0.002**
MAR	+	-0.16	0.04	0.087	NS	**-0.14**	**0.02**	**0.045**
MOO					-	0.20	0.00	0.478
MRM	+	0.31	0.01	0.274				
MUN					-	0.42	0.01	0.179
OLD	NS	0.04	0.00	0.642	-	0.23	0.03	0.132
PEC					-	0.02	0.00	0.774
PIC					+	**0.47**	**0.18**	**<0.001**
PRD					-	**0.49**	**0.29**	**<0.001**
ROA	NS	**-0.51**	**0.41**	**<0.001**	-	**0.51**	**0.29**	**<0.001**
TEL	NS	**-0.33**	**0.11**	**0.009**	-	0.00	0.00	0.900
TUR	+	**-0.71**	**0.59**	**<0.001**	-	-0.04	0.00	0.595
UND					-	**-0.65**	**0.39**	**<0.001**
UNT					+	**-2.66**	**0.41**	**<0.001**
UPN					-	**3.49**	**0.49**	**<0.001**
URM	+	**-1.07**	**0.78**	**<0.001**	-	**0.33**	**0.60**	**<0.001**
WBH					-	**0.52**	**0.39**	**<0.001**

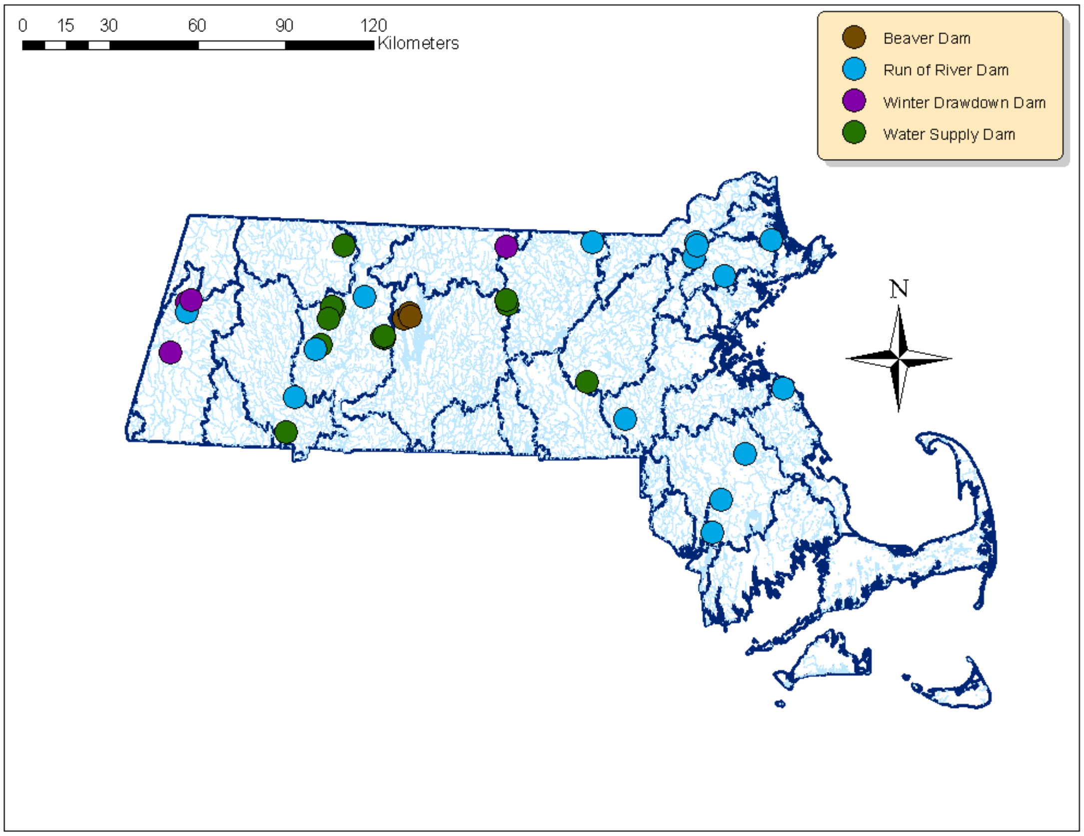

Figure 2.1 – Map of the dams within this study. Dots are color-coded by dam type.

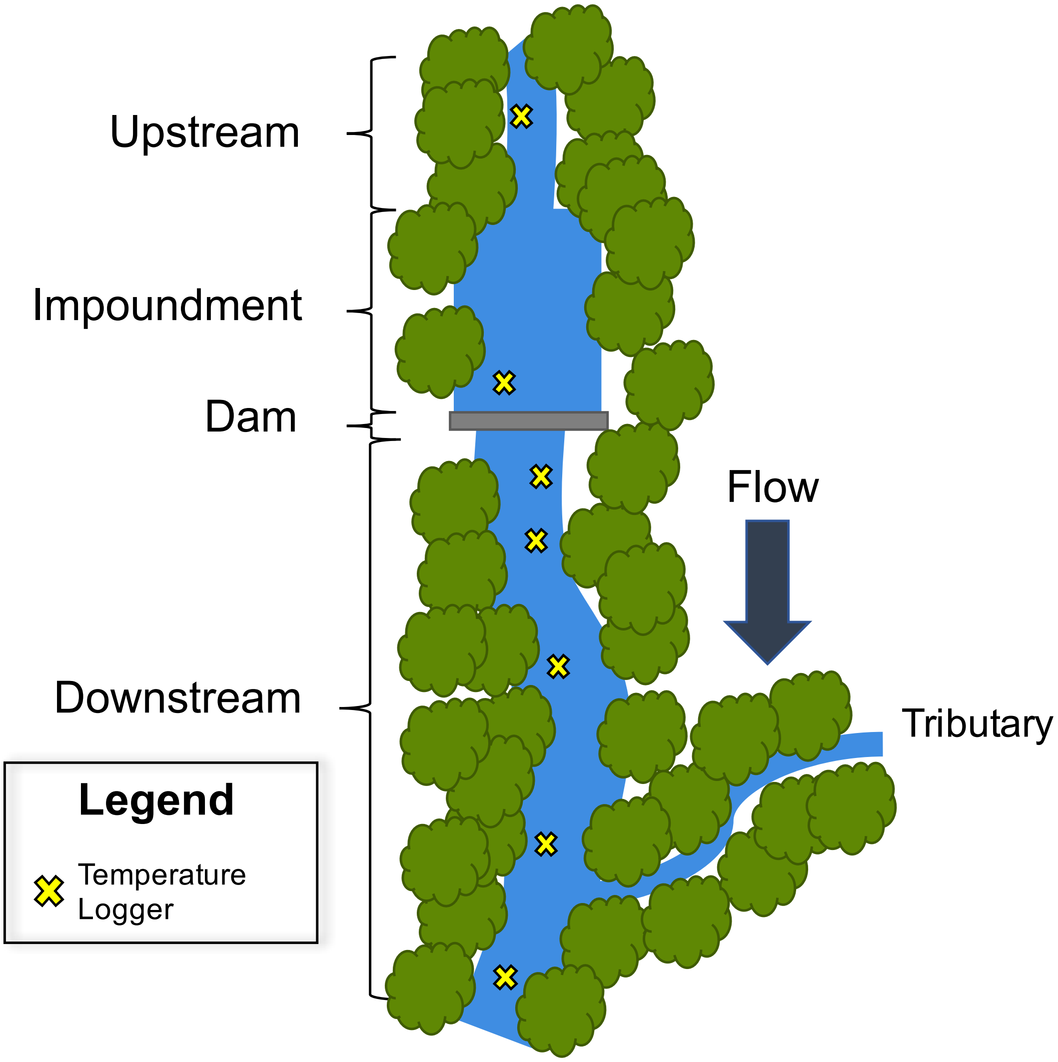

Figure 2.2 – A schematic site design implemented for this study. An upstream logger was deployed (when possible) as a reference condition, an impoundment logger was installed (when possible) to understand the temperature of the water as it was spilling to the downstream reach, and 2 – 6 loggers were deployed downstream under a thermal barrier (e.g., another reservoir, estuary, stream confluence) was reached or until access became prohibitive.

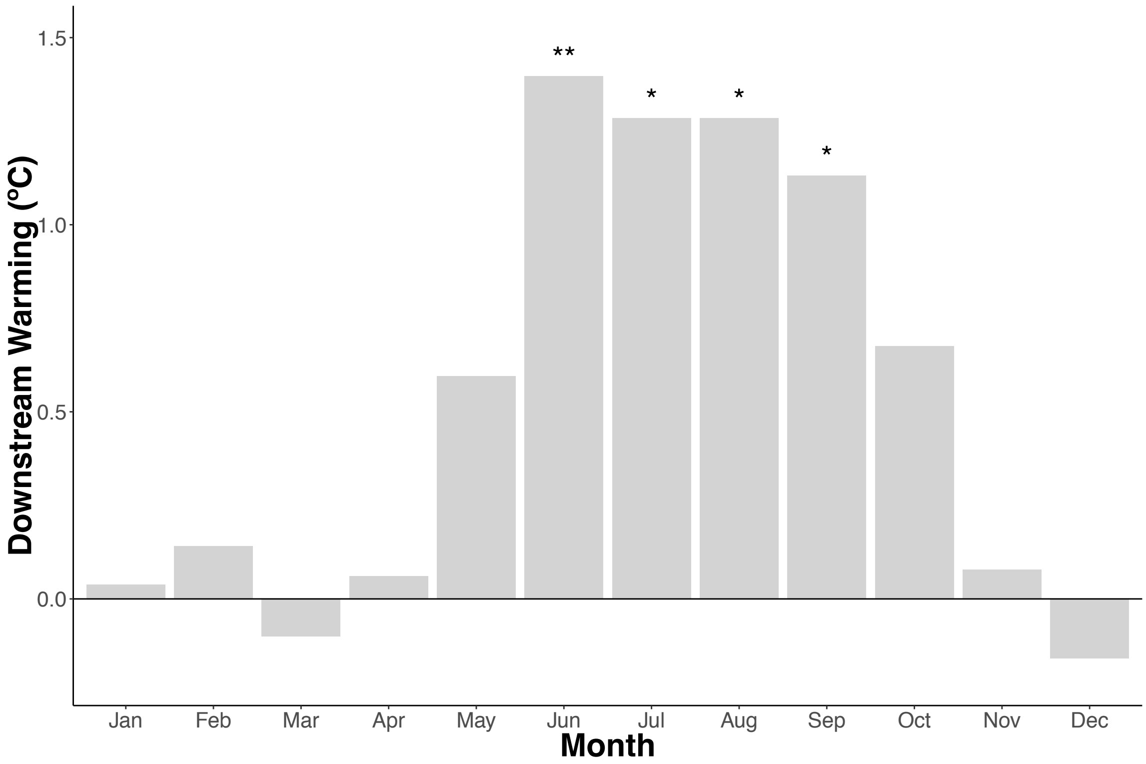

Figure 2.3 – Mean monthly downstream warming magnitudes across all sites cover monthly means from Jun 2014 – Sep 2016. Positive bars indicate warmer downstream temperatures than upstream, negative bars indicate cooler downstream temperatures. Stars represent significant differences from Jan based on an ANOVA of warming magnitude by month. $*p < 0.05$, $**p<0.01$.

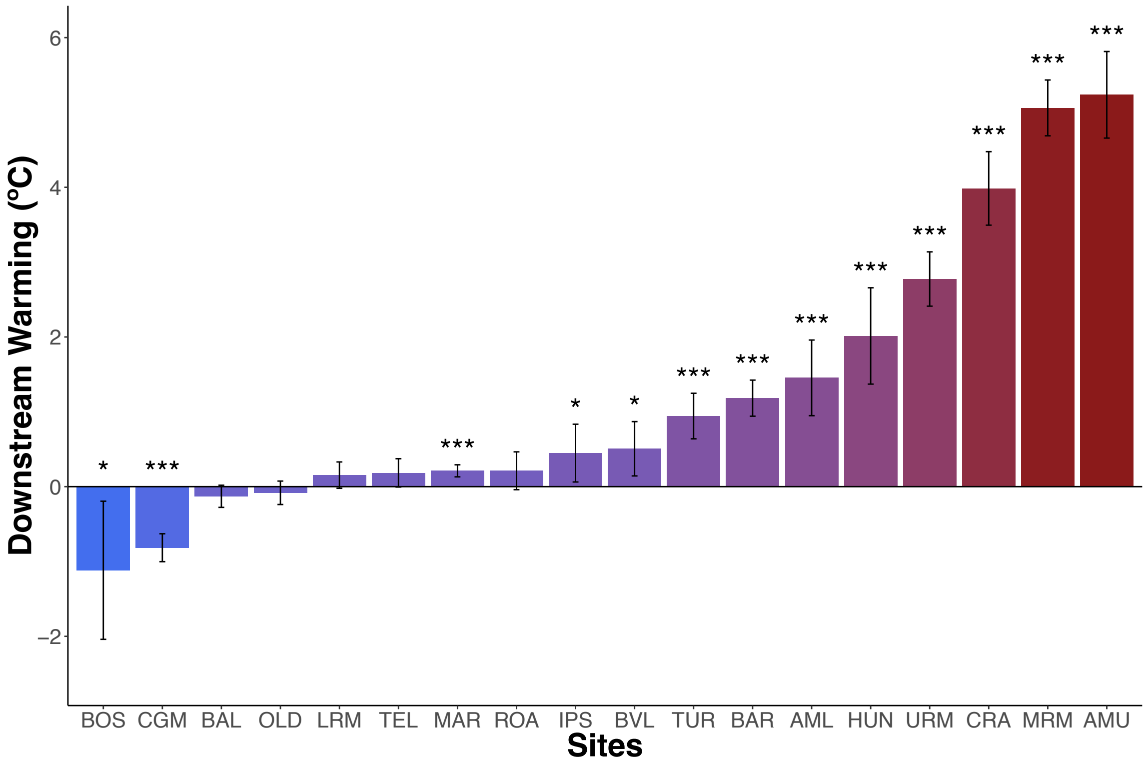

Figure 2.4 – Variation in mean daily August downstream warming by site, ordered from least to most warming. See Table 2.1 for site codes. Positive bars indicate warmer downstream temperatures than upstream, negative bars indicate cooler downstream temperatures. Error bars represent the 95% confidence interval from a paired t-test (by date) comparing downstream and upstream temperatures. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

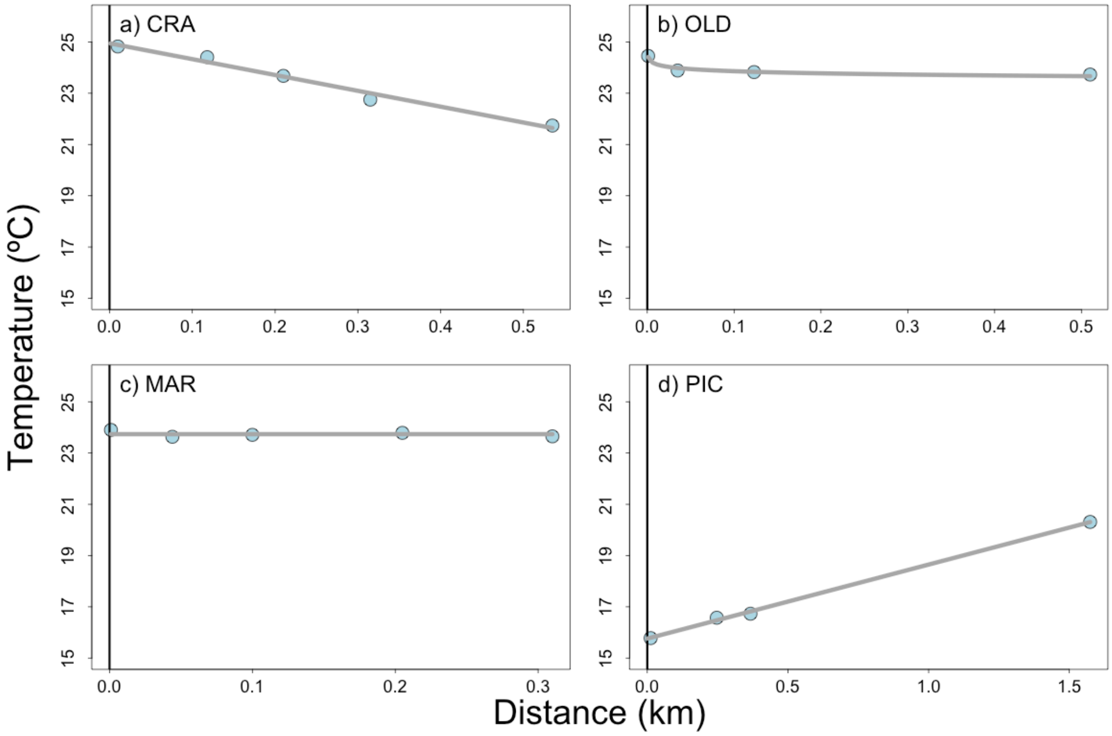

Figure 2.5 – Four different types of decay patterns were observed within this study; (a) linear decay, (b) logarithmic decay, (c) no relationship, and (d) linear increase. See Table 2.1 for site codes.

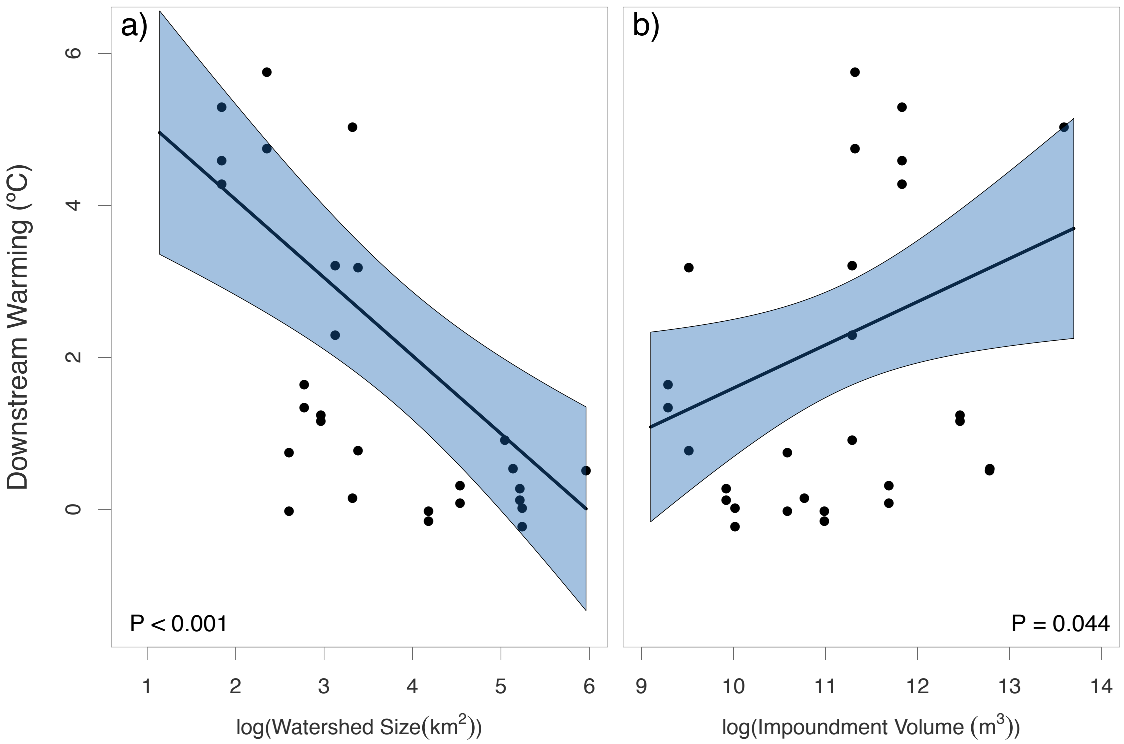

Figure 2.6 – Downstream warming magnitudes were best explained by an additive linear mixed effects model of (a) the log-transformed watershed size and (b) the log-transformed volume of the impoundment with a random effect for site. Dark lines are the mean response for each covariate and shaded polygons represent the 95% confidence interval about that mean (i.e., panel (a) is the relationship between watershed size and downstream warming while holding impoundment volume at its mean).

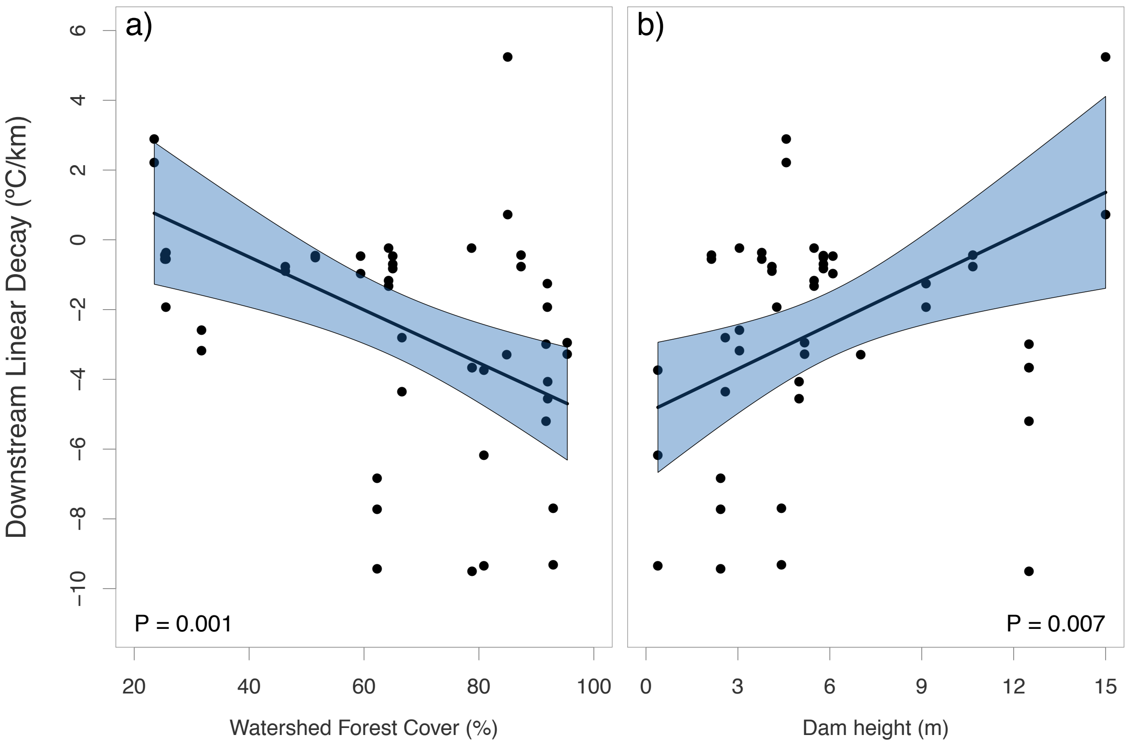

Figure 2.7 – Linear decay rates were best explained by an additive linear mixed effects model of (a) percent forest within the watershed and (b) the height of the dam with a random effect for site. Each panel is the response of that covariate while the other covariate is held at its mean value (i.e., panel (a) is the relationship between dam height and downstream linear decay while holding watershed forest cover at its mean). Dark lines are the mean response for each covariate and shaded polygons represent the 95% confidence interval about that mean (i.e., panel (a) is the relationship between watershed forest cover and downstream linear decay rate while holding dam height at its mean).

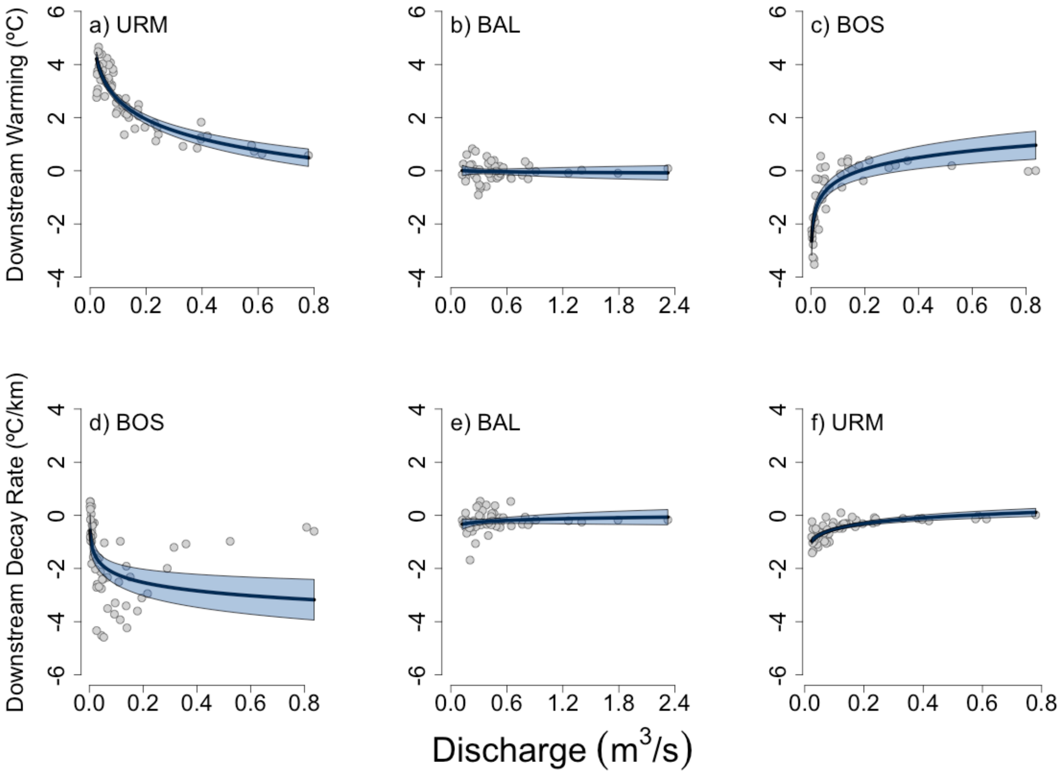

Figure 2.8 – Relationships between flow and warming magnitudes (a-c) were (a) negative, (b) none, and (c) positive; similarly, responses of downstream decay rates (d-f) to discharge were (d) negative, (e) none, and (f) positive. See Table 2.1 for site codes.

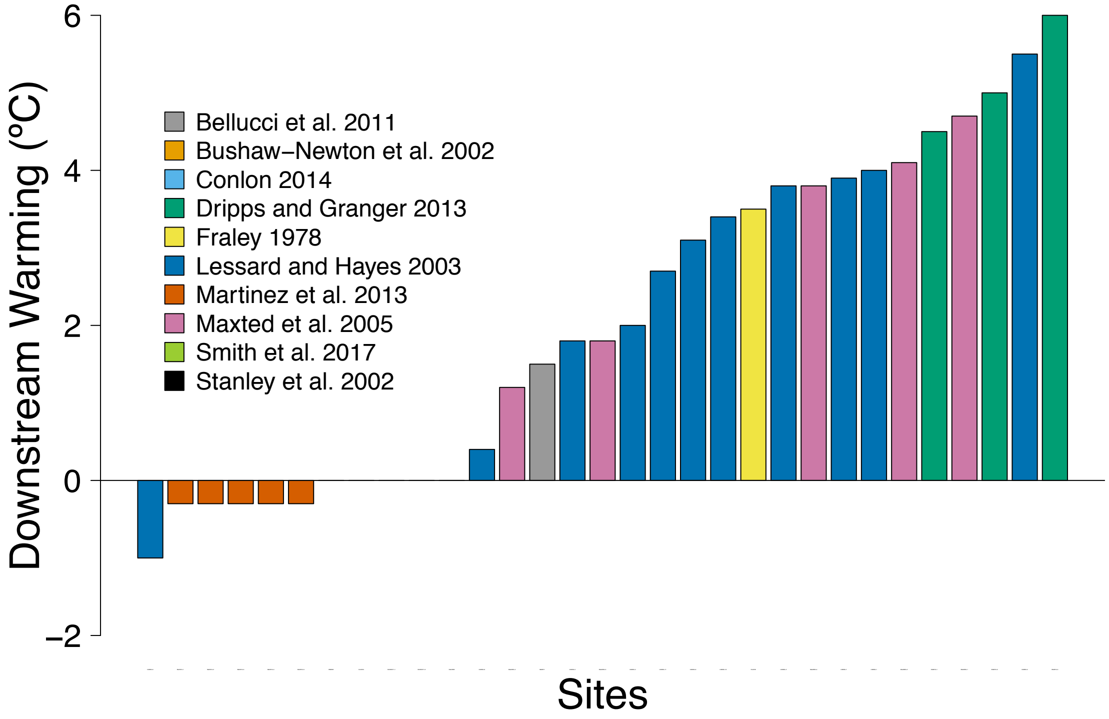

Figure 2.9 – Variation in warming magnitudes downstream of small dams reported across the literature, ordered from least to most warming. Positive bars indicate warmer downstream temperatures than upstream, negative bars indicate cooler downstream temperatures. Bars are color-coded by study (see in-figure legend for studies graphed), and each represents an individual dam site. Several sites (e.g., Bushaw-Newton et al. 2002, Stanley et al. 2002) simply reported no difference (i.e., no number values provided) and are plotted as a bar of 0 °C height.

CHAPTER 3

INVESTIGATING IMPACTS OF SMALL DAMS TO DISSOLVED OXYGEN IN STREAMS

3.1 Introduction

Dams have the potential to alter dissolved oxygen (DO) regimes within a stream, though the direction and magnitude of impact can vary significantly across dam types. Large, bottom-release dams have consistent and predictable impacts to downstream DO concentrations. These dams release water from the bottom hypolimnetic layer of a reservoir, which does not mix with the surface epilimnion layer. This bottom layer of reservoir water has low DO due to the lack of atmospheric mixing, the high demand for oxygen by decomposers, and the decreased rates or lack of photosynthesis in the deeper, darker waters (Bednarek 2001). Reductions in DO have been shown to persist for tens of kilometers downstream of large dams (Bednarek and Hart 2005). The hypoxic pollution of these large dams has been considered such a significant impact that numerous studies have attempted to optimize reservoir operations to minimize the negative effects to downstream DO concentrations (Higgins and Brock 1999, Bednarek and Hart 2005, Jager and Smith 2008).

In contrast to the consistent low levels of DO in the tailwaters of bottom-release dams, the reported impact of small, surface-release dams to DO varies considerably amongst sites. Studies have shown increased (Ignatius and Rasmussen 2016), no change (Bushaw-Newton et al. 2002, Lessard and Hayes 2003), and decreased (Maxted et al. 2005) DO concentrations downstream of small dams. Impacts to the impoundment are more consistently negative, with decreased mean DO (Maxted et al. 2005, Santucci et al. 2005, Ignatius and Rasmussen 2016) and larger diel ranges (Santucci et al. 2005) relative to upstream lotic reaches. Reductions in water flow rates through the impoundment and a more flat-water, lentic habitat decrease turbulence and subsequent atmospheric mixing leading to a reduced concentration of oxygen within the impoundment surface waters. Lentic conditions within the impoundment may foster elevated growth and abundance of macrophytes and algae that, in turn, may increase DO concentrations due to photosynthesis or decrease DO concentrations with algal die-off. The larger surface area of the impoundment, however, may experience increased solar radiation that can lead to warming of the impoundment surface waters (Hamblin and McAdam 2003, Dripps and Granger 2013) and a subsequent decrease in the physical amount of DO the water can hold. This low DO water, as it spills to the downstream reach, could result in decreased downstream DO concentrations, though reaeration may negate this effect to some extent and may be a factor in the sites that did not observe decreased downstream DO.

Dissolved oxygen is fundamental to a number of biogeochemical processes within stream ecosystems, and dams that alter DO regimes within a stream have the potential to impart significant changes on the structure and function of the stream. Oxygen is a key determinant of a stream's metabolic rate, and whether or not a system is a net-producer (autochthonous) or net-consumer (allochthonous) of organic matter (Odum 1956, Cummins 1974, Bernhardt et al. 2017). Alterations to DO concentrations have the potential to shift how the stream processes organic matter and nutrients, and subsequently and how those materials may then be transported to downstream reaches (Allan 1995). Hypoxic impoundments via bacteria found in their anoxic sediments, similar to lakes and wetlands, may serve as hotspots of mercury methylation, the process via which mercury is converted to methylmercury (Watras et al. 1998, Kasper et al. 2012). Methylmercury is a more potent neurotoxin that is highly bioavailable and easily accumulates up the food chain (Watras et al. 1998, Lavoie et al. 2013).

Low DO can impact aquatic organisms not just via ecosystem-level changes or via a potential increased likelihood of biomagnifying methylmercury concentrations, it can also impact aquatic organisms more directly. Decreases in DO below an organism's upper tolerance threshold (incipient limiting tension) result in sub-lethal negative stress effects to organisms. Fish with hypoxic stress experience reduced feeding and growth rates (Dabrowski et al. 2004, Remen et al. 2012, Eliason and Farrell 2014), compromised immune systems (Burt et al. 2013), and reproductive impacts (Wu et al. 2003), which indirectly result in population effects. These hypoxia-induced effects vary among developmental stages of fish (Fuzzen et al. 2011, Elshout et al. 2013), with no consistent hypoxia response patterns across different life stages or body sizes (Doudoroff and Shumway 1970, Davis 1975). For example, Pörtner and Knust 2007 found large body sizes to be most sensitive to hypoxia while Everett and Crawford 2010 concluded that larger body sizes were least sensitive to hypoxia. At the extreme, if DO drops below an organism's incipient lethal tension, the amount of oxygen needed to sustain bodily function is no longer met and the organism will die (Prosser and Brown 1961, Davis 1975).

Impacts to DO can carry significant implications for stream ecosystems and their biota, and with an estimated 2 million dams in the U.S. (Graf 1993), there is a need to better understand how dams impact stream DO. These dam impacts may be most pressing to study within the New England region of the United States, which has the highest density of dams per area in the country (Graf 1999). Most of these dams are small, surface-release structures that were built to power mills during the Industrial Revolution (Walter and Merritts 2008, Magilligan et al. 2016). As these smaller dams have been less studied than larger dams (Lessard and Hayes 2003) and reported results of small dams are variable in their impacts to DO, there exists a major gap in our understanding as to how small dams collectively affect stream DO and, in turn, stream ecosystem structure and function. For this study, I examined how 12 small, surface-release dams across Massachusetts affect stream DO concentrations. Specifically, the objectives were to (1) quantify the impacts of dams to impoundment and downstream DO concentrations relative to upstream, reference reaches and (2) determine the landscape and dam-specific factors influencing impoundment and downstream DO impacts. The results of this study can allow managers to better predict where the most negative impacts to DO can be found across a dammed landscape.

3.2 Methods

3.2.1 Study Area

I assessed the impacts of small dams on temperature at 12 surface-release dam sites in Massachusetts, USA (Table 3.1, Figure 3.1). All 12 sites in this study were former run-of-river mill dams that no longer serve the purpose they were designed for and either have already been removed or are slated for removal as a part of a larger river restoration effort by the Massachusetts Division of Ecological Restoration (MDER). These dams were located across the state, with terrain ranging from mountainous, high gradient slopes in the western part of the state to low gradient coastal plains in the east (MassGIS Digital Elevation Model 1:5,000). Given that the dams were selected opportunistically based on funding, the sites within this study were more distributed along the eastern portion of the state. Despite this, there was still a range of landscape features covered across the sites, as watershed size ranged from 17 – 389 km^2, mean basin forest cover ranged from 25 – 90%, and mean basin impervious cover ranged from 0 – 28%. Landscape features for the 12 sites are detailed in Table 3.2.

3.2.2 Dissolved Oxygen

I used HOBO® Dissolved Oxygen data loggers (U26-001; Onset Computer Corporation, Bourne, MA) that were set to measure DO continuously every 15 minutes. I deployed one logger upstream of the maximum extent of the impoundment, one logger floating just below the surface of the impoundment, and one logger downstream of each of the 12 dams. Loggers were deployed in white PVC flow-through housings to protect the loggers physically and shield them from direct solar radiation (Dunham et al. 2005, Granata et al. 2008). Those in stream reaches were deployed in representative deep pools or runs on rebar, while impoundment loggers were hung from anchored floats ~30 cm below the water surface to capture the DO concentration of water spilling over these epilimnetic-release dams. Impoundment loggers were deployed ~25 m upstream of the dam for safety reasons. I used a multiparameter probe that displayed instantaneous DO readings to determine an installation location representative of each of the three aforementioned reaches.

DO was measured for 1 week in August or September 2015, and for one week during each of the three summer months (July, August, September) in 2016 at each of the dam sites. The only exception to this was Turner Dam (removed in September 2015), where I performed three week-long deployments in August/September 2015 prior to dam removal. I focused on summer months for this study as lower flows (less atmospheric mixing) and higher temperatures (reduced ability to hold oxygen molecules) result in summer months with the lowest DO concentrations throughout the year. These results should represent the worst-case effect of dams on DO.

DO loggers were calibrated to both 100% and 0% saturation prior to their deployment in the field. Logger values were compared to those from a multiparameter sonde at retrieval from the field to ensure biofouling (algal growth on the logger during the deployment period) was not affecting data points. Oxygen data were visually and quantitatively checked using the ContDataQC package in RStudio (RStudio Team 2016) to identify anomalous points. Anomalous data were flagged and removed from analyses.

At the retrieval visit for each impoundment DO logger, I performed a vertical profile of each impoundment >0.5 m deep with a multiparameter sonde. During each vertical profile, temperature, DO (concentration and percent saturation), pH, and conductivity were measured at 0.5 m intervals from the surface down to ~0.5 m above the reservoir bottom. This resulted in each eligible site having 4 vertical profiles taken over the summer months (July, August, September) from the years 2015 and 2016, with the exception of TUR (only 1 profile taken before removal in September 2015), HUN (only 1 profile taken when depth was >0.5m), CGM (not deep enough), and RAT (not deep enough).

3.2.3 Landscape Variables and Dam Features

I used the USGS application StreamStats (http://streamstats.usgs.gov) to delineate the watershed for each dam and calculate landscape variables within that watershed. I generated watershed size, percent forest cover (MassGIS Land Use 2002), percent impervious cover (National Land Cover Database 2011), mean slope, and mean elevation for each watershed. I also calculated reach-scale slope for a 100-m reach centered around the upstream and downstream loggers using Google Earth data (Benson et al. 2014) and determined a ratio of upstream vs downstream slopes.

Impoundment surface area was calculated from the MassGIS MassDEP Hydrography 1:25,000 layer. Dam heights were obtained via technical reports. Impoundment volumes were obtained through the National Inventory of Dams or technical reports, or estimated based on impoundment surface area and max depth (assuming a half-sphere shape). I included a two-level factor (yes/no) for whether or not the dams had some feature (e.g., secondary spillway, fish ladder, significant hole in the dam) present that would allow a high amount of water to pass, based on visual observations in the field.

3.2.4 Data Analysis

I calculated daily minimum, mean, and maximum, and the weekly (i.e., individual deployment) DO concentrations for each reach of each the site in this study. I used paired t-tests (by date) to compare for significant differences between mean impoundment and upstream DO concentrations ("impoundment difference") and to compare for significant differences between mean downstream and upstream DO concentrations ("downstream difference"). I subtracted the daily minimum from the daily maximum to calculate the daily DO range at each of the reaches, and used paired t-tests (by date) to compare differences in daily ranges between the reach reaches for each site. Autocorrelation tests were used to thin daily data to prevent autocorrelation. Two non-sequential days were randomly selected from each deployment such that each site had 8 days of data, with the exception of TUR that had 6 days of data (due to the dam being removed in September 2015).

I used both univariate and multivariate linear regression models to determine how landscape and dam characteristics influence differences in DO response (Objective 2). I regressed the landscape and dam characteristics individually via univariate models against the differences in DO to determine the relative effect of each predictor on both impoundment DO response and downstream DO response. I then ran multivariate linear mixed effects models with the downstream difference and impoundment difference as the response variables and tested all combinations of landscape and dam characteristics as predictors to determine which best explained the variation in response observed across sites. Site and deployment period were tested as random effects to focus on inter-site variation in downstream response. All continuous variables were Z-score standardized to eliminate significance due simply to differences in the scales of units amongst predictor variables. I used both pairwise correlations and variance inflation factors (VIF) to examine collinearity among predictor variables and did not include predictors with correlation coefficients >0.7 and VIF values >3 in these models. Akaike information criterion corrected for small sample size (AIC_c; Burnham and Anderson 2002) was used to determine the best supported model.

I used R version 3.3.1 (R Core Team 2016) to perform all analyses. Results were considered significant if $p < 0.05$.

3.3 Results

3.3.1 Impacts to DO Concentrations

Most lotic reaches had mean deployment DO values >5.0 mg/L, with the exception of 3 upstream reaches and 4 downstream reaches (Figure 3.2). Three impoundments also experienced at least one weeklong deployment with mean DO <5.0mg/L. The threshold of 5.0 mg/L is used by the Massachusetts Department of Environmental Protection as a concentration below which waters may be considered impaired for DO. Upstream reaches averaged 7.10 mg/L with mean diel ranges of 1.85 mg/L/day, while downstream reaches averaged 6.40 mg/L with average diel ranges 1.28 mg/L/day. Impoundment surface conditions averaged significantly reduced mean daily concentrations of 6.11 mg/L, with average diel ranges of 2.61 mg/L/day.

Comparing these reaches within sites, impoundments had lower daily mean DO concentrations compared to their upstream reference reaches at 8 of 12 sites (Figure 3.3). Three sites showed no difference between upstream and impoundment DO, including one site that had been dewatered for several years (CGM). One site had elevated mean daily impoundment DO relative to the upstream reach. Across the 12 sites, impoundment DO concentrations were an average of 1.10 mg/L lower (range= -4.01 to +2.14 mg/L) than upstream reference reaches.

The decreases in impoundment DO did not translate to a widespread decrease in downstream DO concentrations. Downstream DO concentrations were significantly lower than upstream at 4 sites, and were significantly elevated relative to the upstream condition at 3 sites (Figure 3.4). Five sites showed no difference between upstream and downstream conditions. Across the 12 sites, the downstream DO concentrations were on average 0.72 mg/L lower (range= -8.13 to +2.13 mg/L) than upstream reaches.

Most sites (7 of 12) experienced similar diel DO variation in the impoundment and upstream reaches (Table 3.3). Four sites experienced larger daily ranges in the impoundment relative to the upstream reach, and one site had a slightly reduced daily range within the impoundment compared to the upstream reach. Most sites (7 of 12) experienced suppressed diel variation (more stable daily DO concentrations) downstream of the dam relative to upstream. Two sites had no difference between upstream and downstream daily ranges and three sites had increased daily ranges downstream relative to upstream (Table 3.3). Because CGM had been dewatered and did not have a dam effect, it was not included in further analyses.

All three of the impoundments that experienced at least one mean deployment concentration <5.0 mg/L experienced water column hypoxia (DO concentrations <2.0 mg/L). Two sites (BAR and HUN; maximum impoundment depths of 1.1 and 0.7 m, respectively) experienced hypoxic surface conditions throughout the water column. The third site (BOS; maximum impoundment depth= 2.2 m) displayed hypoxic conditions near the impoundment bottom as the surface DO (average= 4.63 mg/L) decreased to <2.0 mg/L (average= 1.59 mg/L) ~0.5 m above the impoundment sediments (Table 3.4). In addition, a fourth site (URM; maximum impoundment depth= 4.6 m) that did not experience mean deployment surface DO concentrations <5.0 mg/L consistently experienced complete impoundment summer stratification, whereby a vertical profile with a multiparameter sonde recorded not just hypoxic conditions, but anoxic conditions (DO= 0 mg/L) within the water column. The average depth of stratification across 4 vertical profiles was 2.4 m, suggesting that ~50% of the impoundment water column (at the deepest point) was anoxic throughout the summer. The remaining 6 sites with impoundments deep enough for a vertical profile exhibited little to no summer stratification across the 4 profiles (Table 3.4).

3.3.2 Drivers of Inter-Site Variation

The output of the univariate models indicates that only two landscape predictors significantly predicted the difference between downstream and upstream DO concentrations – the upstream reach slope and the ratio between the upstream and downstream slopes (Table 3.5). Both predictors had a negative relationship with downstream DO difference, indicating that sites with steeper upstream slopes had a greater downstream DO loss than those with lower gradient upstream reaches. None of the characteristics of the dam were significant predictors of downstream DO change individually. Multivariate modeling indicated that the difference in downstream DO relative to upstream was best supported by the model with upstream slope, impoundment volume, and a random effect for site (Table 3.6). Both upstream slope and impoundment volume were negatively related to the difference in downstream DO, indicating that sites with steeper upstream basins and larger volumes of impounded water behind the dam experienced the greatest losses of downstream DO. However, the more parsimonious model consisting of just upstream slope and a random effect for site was equally as plausible ($\Delta AIC_c = 0.1$). While upstream slope and impoundment volume were the top supported models consisting of landscape and dam characteristics, the best predictor of downstream DO response was the change in impoundment DO relative to upstream.

The change in impoundment DO relative to upstream was significantly affected by two univariate models: the upstream slope (negative) and the watershed size (positive; Table 3.5). Similar to the downstream response, sites with higher gradient upstream reaches and those located within smaller watersheds experienced the greatest decrease in impoundment DO relative to upstream DO. For the linear mixed effects models, the variation in magnitude of impoundment DO difference relative to upstream DO was best explained by a model that included upstream slope (negative), basin size (positive), dam height (positive), and a random-intercept term for site (Table 3.6). Given the directionality of these relationships, dams with steeper upstream slopes and shorter dams, and those located in smaller watersheds experienced the greatest decrease in impoundment DO relative to upstream reference reaches. There were several other models that were equally as plausible ($\Delta AIC_c < 2.0$) as this top model, which all contained the same variables, just in varying combinations (Table 3.6).

3.4 Discussion

3.4.1 Effects of Dams on DO

Not surprisingly, given the conversion from a lotic to a lentic waterbody type, the impoundment formed by damming experienced the greatest decreases in DO concentrations, with 66% of sites in this study exhibiting decreased mean daily DO within the impoundment relative to their upstream reach. These observations support results from previous studies in which impoundments experienced the largest decreases in DO concentrations in a dammed stream reach (Maxted et al. 2005, Ignatius and Rasmussen 2016). The slow-moving water within impoundments facilitates the accumulation of fine sediments and organic matter (Petts 1984, Stanley and Doyle 2002). The decomposition of these organic sediments (and the necessary consumption of DO in the process) has been implicated in past research as the driver for decreases in impoundment DO (Maxted et al. 2005). Decreases in impoundment DO may also be due to the physical inability of warm impoundment surface waters (resulting from increased surface area and decreased canopy cover) to hold as much oxygen as cooler waters. Additionally, the flat-water impoundment may receive less oxygen to the stream system as a result of reduced surface water turbulence and associated reaeration (Raymond et al. 2012).

Over half of the sites in this study did not show a difference between upstream and impoundment mean diel range magnitudes. Four sites, however, did experience increased diel variation in the impoundment relative to upstream reaches, supporting results observed previously in the literature (Maxted et al. 2005, Santucci et al. 2005). The decrease in flow rates through an impounded reach can minimize the shear forces that would otherwise break up algal communities in free-flowing riverine environments, which in turn, can create favorable conditions for high algal growth within the impoundment (Soballe and Kimmel 1987). High rates of oxygen production during photosynthesis and oxygen consumption via respiration by high concentrations of autotrophs would lead to the large daily fluxes in oxygen observed within impoundments. Most of the sites in this study, based on qualitative observations from the field, did not appear to be significantly more eutrophic than their upstream reference reach, and may be the reason that only four sites experienced elevated daily ranges within the impounded reach. These qualitative observations are supported by reach-scale chlorophyll samples taken at a subset (n= 5) of these 12 sites that occurred outside of the study period. The results from this one-time sampling event indicate that there was no consistent or significant trend in chlorophyll concentrations between reaches across the sampled sites, with only 1 site having the highest chlorophyll concentrations at that site found within the impoundment.

The decreased DO concentrations of the eight impoundments translated to decreases in mean daily downstream DO at four dams (25%) in this study. I predicted that the directionality and magnitude of impact within the impoundment would determine the directionality and magnitude of impact in the downstream reach. Likely due to the reaeration of the impoundment water as it spills over the dam to the downstream reach, these small, surface-release dams appear to be able to self-recover negative impacts to impoundment DO in the downstream reach, if the impact is small enough. In other words, to translate a negative DO effect to the downstream reach, DO loss needs to be greater than the amount of DO recovered during spilling. This likely mechanistic effect can be seen across a number of the sites in this study, and considering their reach-scale impacts and features of the stream and dam. Consider the sites with decreases in impoundment DO (Figure 3.3), which can be broken into two groups of relatively similar decreases in impoundment DO; a low-impact group consisting of RAT, MAR, TEL, BAL, and TUR, and a high-impact group consisting of URM, BAR, and HUN. Starting with the low-impact sites, the three with high turbulence in their spillways (MAR, TEL, TUR; MAR had a rough boulder spillway, while TEL and TUR had auxiliary spillways via which much of the river flow was able to bypass the dam structure) did not have decreased downstream DO; the turbulence associated with spilling recovered the DO lost in the impoundment. BAL, by contrast, maintained decreased DO in the downstream reach, which may be due to the short, low-angle spillway that did not appear to cause much turbulence as water spilled to the downstream reach in the summer. Within the high-impact group, URM and BAR maintained decreased DO within the downstream reach, while the large DO losses within the impoundment at HUN did not carry over to the downstream reach. The downstream reach at HUN was very high gradient, and may have introduced a large amount of oxygen via turbulence in the stream; the downstream reach also experienced a slight tidal effect, and DO losses may have been abated by the introduction of a separate water source. The water levels within the low-impact impoundment at RAT dropped below the spillway each summer and resulted in a downstream reach that was comprised of standing pools of water, with no oxygen within them.

The spilling water that helped to abate impoundment DO losses in the downstream reaches at half of the impacted sites may also have contributed to the decreases in diel variation observed at over half of the downstream study reaches. Relative to upstream reaches, Maxted et al. (2005) observed elevated diel DO ranges within the impoundments relative to upstream reaches. As water spilled over the dams, the high variation in diel range was carried downstream, though the magnitude of change was not as large as was observed within the impoundment. A similar trend was observed at two of the sites in this study (BAR and URM), and likely the reason it was only observed at these two was the fact that most of these sites did not experience increased impoundment diel variation relative to their upstream reach. The constant flow of water over the spillway and the constant aeration from that process seems to then have dampened diel variability in downstream reaches relative to their upstream reference reaches. Studies have observed a reduction in the diel variation of both temperature and DO downstream of hydroelectric dams relative to their upstream reaches (Lowney 2000, De Baets 2016), though there does not appear to be any evidence of this muted diel DO variation below small, low-head dams in the literature.

Two of these four sites experienced mean daily values >5.0 mg/L upstream of the dam that decreased to values <5.0 mg/L downstream of the dam, suggesting the dam may be impairing the downstream water for oxygen.

3.4.2 Factors Explaining DO Changes

The slope of a dam's upstream reach was the most prevalent predictor of reductions in impoundment DO change relative to upstream conditions. Streambed slope is tightly correlated with turbulence and reaeration rates, such that higher gradient slopes experience greater turbulence and higher rates of reaeration into the stream (Raymond et al. 2012, Benson et al. 2014, Hall et al. 2016). The sites in this study that had the most reduced DO levels had the highest concentrations of DO in their upstream reaches and had the largest potential for impact. By contrast, low gradient upstream reaches with naturally lower reaeration rates should have a decreased potential to be affected by a reduction in flow and reaeration rates caused by a dam's backwatering effect. The significant effect of basin size on decreases in DO likely reflects a slight overlap in the underlying mechanism of impact as upstream slope. Basin size and upstream slope were slightly correlated, suggesting that many of the highest gradient upstream slopes were found in smaller basins. However, smaller basins also have cooler, headwater streams that physically can hold more DO than warmer temperature streams. Sites with smaller upstream basins likely represent greater potential for impact, due both to naturally higher DO concentrations within the cooler streams and potentially due to higher turbulence in stream slopes.

Shorter dams had larger decreases in impoundment DO that likely were the result of high levels of decomposition of plant matter within these productive impoundments. Warm temperatures and high light availability throughout the shallow water column, coupled with the reduction in shear forces, made the shallow impoundments behind these short dams highly productive environments with high concentrations of aquatic macrophytes and algae (Soballe and Kimmel 1987). In early summer, these plant communities, through high rates of photosynthesis, can produce conditions of hyperoxia, or oxygen supersaturation, that is followed in late-summer by hypoxic conditions caused by the aerobic decomposition of algae and macrophytes by microbe communities (Boesch et al. 2001, Mallin et al. 2006, Shen et al. 2014).

The best landscape predictor of downstream DO difference was the ratio between upstream and downstream slopes, suggesting there may be a legacy effect that explains the wide variation observed across these sites. In their site selection, dam builders frequently chose to construct dams at the site of a natural and existing rapid change in channel slope, particularly in lower gradient, coastal rivers. These existing features (e.g., bedrock outcrops, falls, etc.) likely had a downstream riffle with naturally higher DO concentrations due to their aerating properties (Benson et al. 2014). As such, the downstream reaches of several of these dams (e.g., Bostik, Old Mill) have larger and more prominent riffles than those found upstream. That these sites also had higher DO concentrations downstream of the dam than upstream may represent a natural condition of the stream (i.e., the downstream reach would have higher DO regardless of the presence of the dam) and not one in which the dam is having a beneficial impact on downstream DO conditions.

3.4.3 Management Implications

The decreases in DO observed within this study did not spell disaster for aquatic organisms in Massachusetts, but several sites did experience impacts that may carry negative biotic implications. Across the 12 sites investigated within this study, four had significant decreases on mean daily downstream DO concentrations. Three of these four sites decreased downstream DO below a state threshold (5.0 mg/L) that may result in elevated stress for aquatic organisms. One of those downstream reaches, however, was <5.0 mg/L because water levels within the impoundment dropped low enough throughout the summer to prevent spill, and thus the downstream reach was comprised of standing pools of anoxic water, and thus may represent a case in which the downstream impairment is not limited to DO, but the fact that the dam cut off the downstream reach from flow and the remainder of the stream. In addition to these threshold and potential biotic implications, four (BAR, BOS, HUN, URM) of the eight impoundments that experienced DO losses had vertical profiles that suggested that hypoxia and anoxia may be a problem in bottom waters and impoundment sediments. Anoxic sediments within these impoundments have the potential to serve as hotspots for mercury methylation, which as the more bioavailable form of the toxic metal, present a real threat to aquatic life and terrestrial organisms (including humans) that may be consuming these aquatic organisms.

The 12 dam sites studied within this project are all dams that either have already been removed or are being considered for removal in the future, and tie these results back into the practice of river restoration and dam removal. The results ascertained herein suggest that sites with high gradient upstream reaches faced the greatest DO losses, likely given that they experience the largest change in habitat structure in conversion to a flat-water, lentic impoundment behind a dam. Additionally, shallow impoundments behind short dams appear to be most susceptible to high rates of establishment of aquatic macrophytes and algae, whose breakdown consumes large amounts of oxygen in late summer. The sites that experienced the greatest losses to impoundment DO were in general the sites that had the greatest losses in downstream DO, relative to their upstream reference reaches. For practitioners looking to improve water quality and eliminate DO impairment by dams, efforts should focus on sites with high gradient upstream reaches and shallow, eutrophic impoundments, and those within small basins as the sites that will likely experience the greatest improvements to DO regimes following dam removal.

As much of this discussion focused on the ability of dams to self-recover impoundment DO losses via turbulence as water flows over the spillway to the downstream reach, it would seem reasonable to wonder if removing dams might lead to a net decrease as that spillway is lost with the dam. Reaeration is determined largely by bed slope and flow rates, and as dams are removed, flow through downstream reaches may be increased. Additionally, many dams were built at a point on a stream at which there was an abrupt change in bed slope and/or a natural constriction at which maximum storage could be achieved with minimal construction. This pattern was observed at one site that was removed within this study (TUR), as the dam had been built atop a cascade that had been cut off from summer flows while the dam was in place within the stream. Separately, in many dam removals, especially those with surrounding infrastructure, additional activities are required to achieve separate project goals (e.g., constructing a riffle to prevent increased shearing and undercutting forces on an upstream bridge's abutments). The construction of such a riffle at the former dam site (as observed at the former dam site MAR within this study) can help to maintain the turbulence and reaeration rates accomplished via the dam's spillway.

3.4.4 Future Considerations

Past research on the impacts of small, low-head dams on DO concentrations were limited, both in study number and the number of dams investigated within each study. The 12 sites within this study was a large improvement in the number of study sites over past literature and allowed for comparison of differences across sites and to investigate drivers of differences in impact. With several head of tide sites that received tidal water on their downstream face, and one site with a large wetland complex upstream, this study or future studies, could benefit from more study sites (i.e., dams). Additionally, while the landscape features within the watersheds of these 12 sites varied considerably, most sites were concentrated within the eastern portion of the state in the coastal lowlands, and a more even distribution across the state and across landscape (e.g., elevational) features would allow greater insights into the mechanisms at play. The study design at each site (a single upstream, impoundment, and downstream logger) made it feasible to confirm that changes to the impoundment serve as a mechanistic predictor of downstream impacts (in the absence of external features such as auxiliary spillways, tidal influence, etc.). Having longer deployments (i.e., longer than a week at a time, and extending further into the spring and fall) would make it possible to better understand the impact of high flow events that have a hypothesized homogenizing effect on DO concentrations throughout the study reach. Summer conditions did not vary much across deployments (i.e., summer months), with the exception of relationships observed within the impoundment and downstream of those dams with highly productive impoundments, which showed that worse-case conditions occurred from ~mid-August to ~mid-September. Future investigations into the impact of small dams on DO concentrations that simply aim to understand worse-case scenarios may be best off to just focus on this time period. However, if the researcher aims to understand broader impacts to DO by dams, extending the study period into the spring and fall would be appropriate.

While these results suggest that small dams are not having catastrophic impacts on stream DO, there were several instances in which dam-induced impacts may carry negative implications for biota and streams. Given the prevalence of dams across the landscape, even a couple dams within this study of 12 having impacts suggest that there are thousands of dams across the landscape that may be impairing stream ecosystems for DO.

Table 3.1 – Features of the dams analyzed within this study.

Name	Site Code	Stream Name	Dam Height (m)	Impoundment Surface Area (km^2)	Impoundment Volume (m^3)	Impoundment Auxiliary Spillway
Balmoral	BAL	Shawsheen R.	2.1	0.023	22402	No
Barstow's Pond	BAR	Cotley R.	2.6	0.041	259030	No
Bostik / South Middleton	BOS	Ipswich R.	3.1	0.075	77709	No
Cotton Gin Mill	CGM	Satucket R.	1.5	0.001	200	No
Hunter's Pond	HUN	Bound Bk.	3.4	0.005	13568	No
Ipswich Mills	IPS	Ipswich R.	3.2	0.131	356428	Yes
Marland Place	MAR	Shawsheen R.	3.8	0.021	20352	No
Old Mill	OLD	Charles R.	4.1	0.031	59207	Yes
Rattlesnake Brook	RAT	Rattlesnake Bk.	1.2	0.015	18300	No
Tel-Electric	TEL	W.B. Housatonic R.	6.1	0.044	119481	Yes
(Millie) Turner	TUR	Nissitissit R.	3.1	0.069	80176	Yes
Upper Roberts Meadow	URM	Roberts Meadow Bk.	10.7	0.017	80176	No

Table 3.2 – Landscape characteristics of the dams analyzed within this study. See Table 3.1 for site abbreviations.

	Watershed					Reach	
Site	Area (km^2)	% Forest	% Impervious	Mean Elevation (m)	Mean Slope (%)	Upstream Slope (%)	Downstream Slope (%)
BAL	188.8	25.3	27.6	44.5	4.2	0.9	0.2
BAR	19.4	66.6	11.9	23.8	3.0	7.4	0.4
BOS	113.4	31.7	20.6	32.9	4.1	0.0	0.0
CGM	55.4	41.2	14.9	28.5	2.7	0.7	0.1
HUN	29.5	72.5	3.2	29.9	4.8	0.8	4.0
IPS	388.5	49.5	12.7	30.2	5.1	0.0	0.1
MAR	183.9	25.5	27.4	44.5	4.2	1.4	0.1
OLD	65.5	46.3	21.3	95.1	5.4	0.2	0.3
RAT	17.1	89.7	1.3	47.9	6.7	0.8	0.1
TEL	93.5	59.4	4.5	432.8	13.3	0.1	0.5
TUR	155.1	78.7	1.5	136.6	8.0	1.3	0.5
URM	22.8	87.3	0.2	281.0	12.7	5.6	0.7

Table 3.3 – Results of paired t-tests (by date) comparing daily DO ranges between downstream (DS), upstream (US), and impoundment (IMP) reaches at the 12 study sites. Estimate reflects the difference between the first reach relative to the second reach (i.e., DS-US estimate of -0.39 reflects a decrease of 0.39 mg/L/day in diel range downstream of the dam relative to upstream). See Table 3.1 for site abbreviations. Bold estimates are significant ($p< 0.05$).

Site	Comparison	Estimate	T	P
BAL	DS-US	**-0.39**	-2.63	0.034
	IMP-US	**0.73**	3.01	0.020
BAR	DS-US	**2.03**	3.62	0.008
	IMP-US	**3.81**	5.75	<0.001
BOS	DS-US	0.26	0.48	0.648
	IMP-US	-0.16	-0.48	0.643
CGM	DS-US	**-0.93**	-2.50	0.047
	IMP-US	-0.48	-2.21	0.069
HUN	DS-US	**-1.49**	-2.73	0.029
	IMP-US	-0.67	-0.72	0.496
IPS	DS-US	**6.54**	6.07	<0.001
	IMP-US	**-0.76**	-2.28	0.044
MAR	DS-US	**-2.12**	-9.35	<0.001
	IMP-US	-0.18	-0.70	0.508
OLD	DS-US	**-0.85**	-3.17	0.013
	IMP-US	0.27	0.92	0.383
RAT	DS-US	**-0.67**	-5.97	<0.001
	IMP-US	1.56	2.00	0.101
TEL	DS-US	**-0.26**	-2.41	0.047
	IMP-US	**4.33**	6.65	<0.001
TUR	DS-US	-0.07	-0.68	0.523
	IMP-US	0.20	1.70	0.165
URM	DS-US	**0.27**	3.58	0.007
	IMP-US	**2.07**	5.18	<0.001

Table 3.4 – Average profile length (i.e., depth covered by vertical profile measured from surface down), average surface DO, average bottom DO, and the average difference (surface – bottom) in DO between the impoundment surface and bottom at the time of the profile.

			DO (mg/L)		
Site	Max Depth (m)	Profile Length (m)	Surface	Bottom	Difference
BAL	1.6	1.3	6.18	4.98	1.20
BAR	1.1	0.5	5.43	3.92	1.51
BOS	2.2	1.6	4.63	1.59	3.04
CGM	0.6	NA	NA	NA	NA
HUN	0.7	0.4	1.66	0.75	0.92
IPS	1.9	1.3	6.51	4.67	1.84
MAR	3.0	2.3	6.44	5.31	1.13
OLD	1.4	0.8	5.58	5.10	0.48
RAT	0.3	NA	NA	NA	NA
TEL	2.1	1.5	4.92	4.95	-0.03
TUR	1.6	1.0	8.07	8.07	0.00
URM	4.6	3.9	6.26	0.00	6.26

Table 3.5 – Standardized estimates for univariate models between watershed characteristics and dam features and downstream difference (downstream – upstream) and impoundment difference (impoundment – upstream) mean daily DO concentrations. Note that all estimates are scaled relative to their means to allow for equal comparison of effect size across all parameters. NA = variable not applicable (circular). Bold font indicates significant differences at $p < 0.05$, and '†' signifies predictor variables that were log-transformed for analyses.

	Parameter	*Downstream difference*	*Impoundment difference*
Basin Characteristics	Basin forest cover	-0.45	-0.74
	Basin impervious cover	0.32	0.50
	Mean basin slope†	0.03	0.37
	Upstream slope	**-0.60**	**-0.89**
	Downstream slope	-0.23	NA
	Upstream/downstream slope	**-0.61**	**-0.89**
	Mean basin elevation†	0.09	0.35
	Basin size†	0.37	**1.04**
Dam Features	Dam height†	-0.03	0.33
	Reservoir surface area	0.21	0.82
	Reservoir volume	-0.27	0.01
	Auxiliary spillway	0.57	1.13
	Impoundment – upstream DO difference	**0.84**	NA

Table 3.6 – Model output from the top models (ΔAIC< 2.0) explaining the variation in changes, relative to upstream concentrations, of the downstream reach ("downstream difference") and the impoundment ("impoundment difference"). All models include site as a random intercept term.

Model	AIC_c	ΔAIC_c	R^2
Downstream difference			
Upstream Slope + Volume	115.5	0.0	0.46
Upstream Slope	115.6	0.1	0.38
Impoundment difference			
Upstream Slope + Basin Size + Dam Height	146.7	0.0	0.48
Upstream Slope	147.1	0.3	0.32
Upstream Slope + Basin Size	147.8	1.1	0.37
Upstream Slope + Dam Height	147.8	1.1	0.36
Basin Size	148.1	1.4	0.26

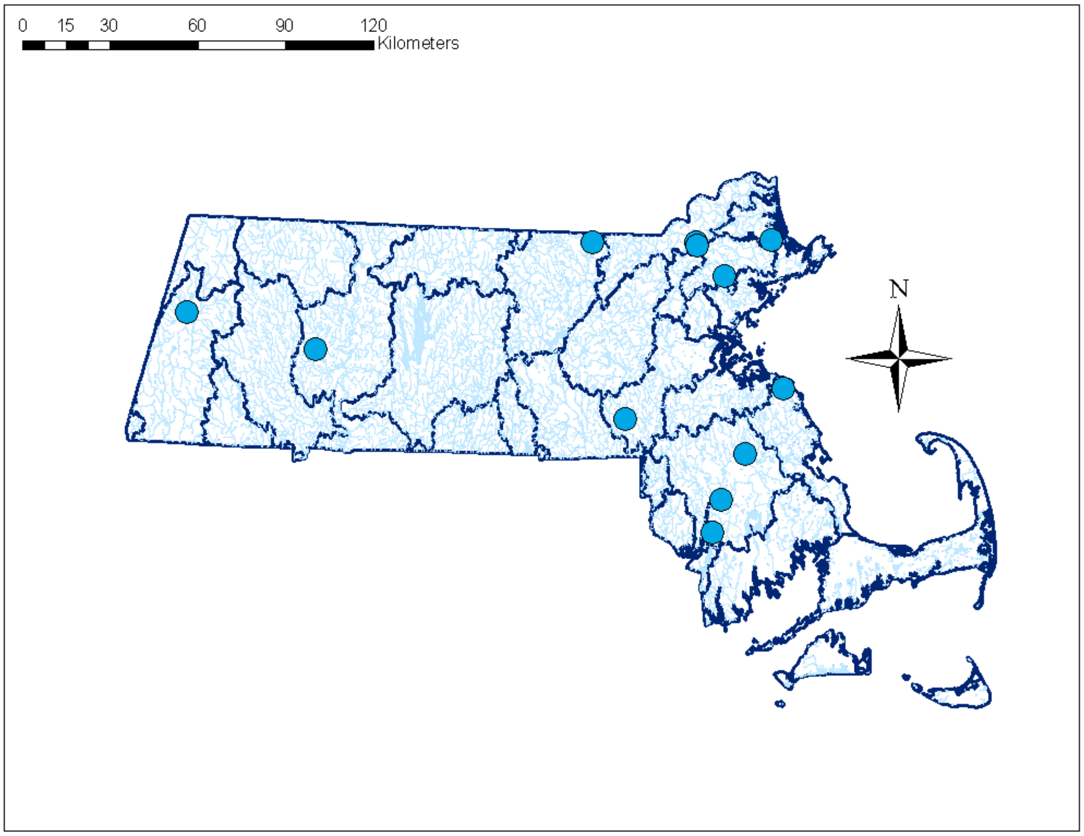

Figure 3.1 – Map of the dams within this study.

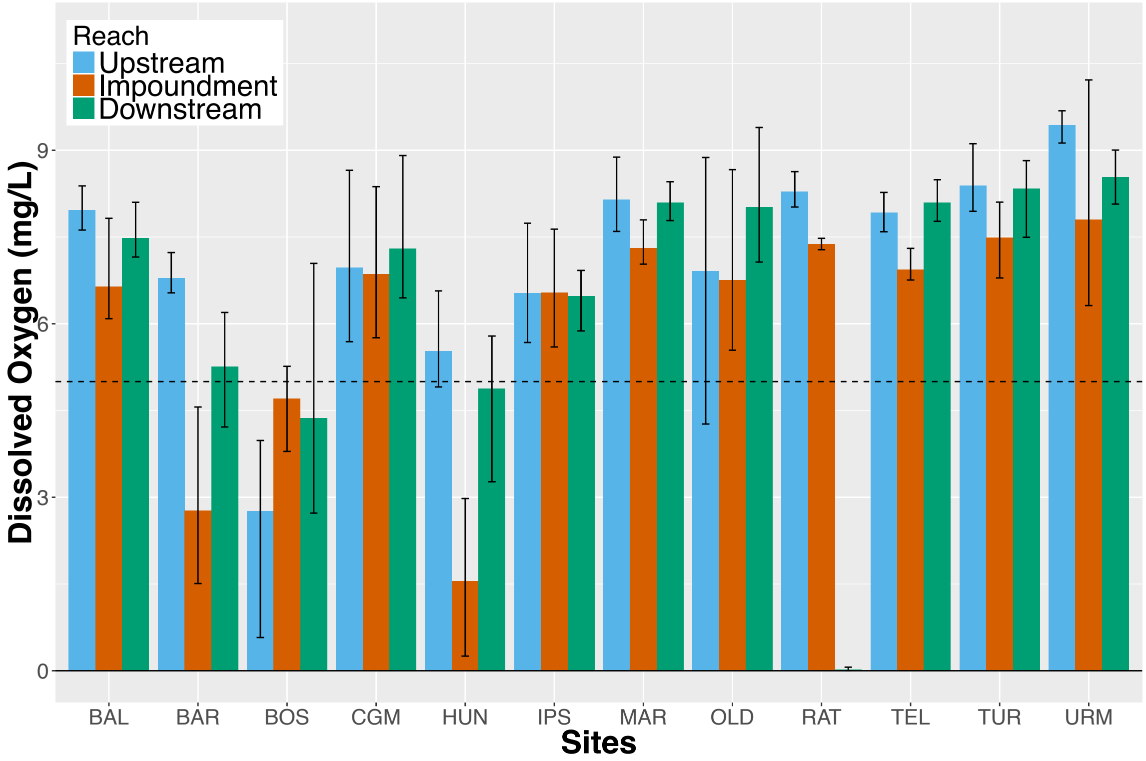

Figure 3.2 – Mean values for each reach at each of the 12 study sites with error bars extending from the minimum of the 4 deployment mean concentrations to the maximum of the 4 deployment mean concentrations. Turner (TUR) only had 3 pre-removal deployments, and so the bars at this site extend from the minimum to maximum of the 3 deployment values. The dashed line is at 5.0 mg/L of DO, a threshold used by the Massachusetts Department of Environmental Protection, below which waters may be considered impaired for DO.

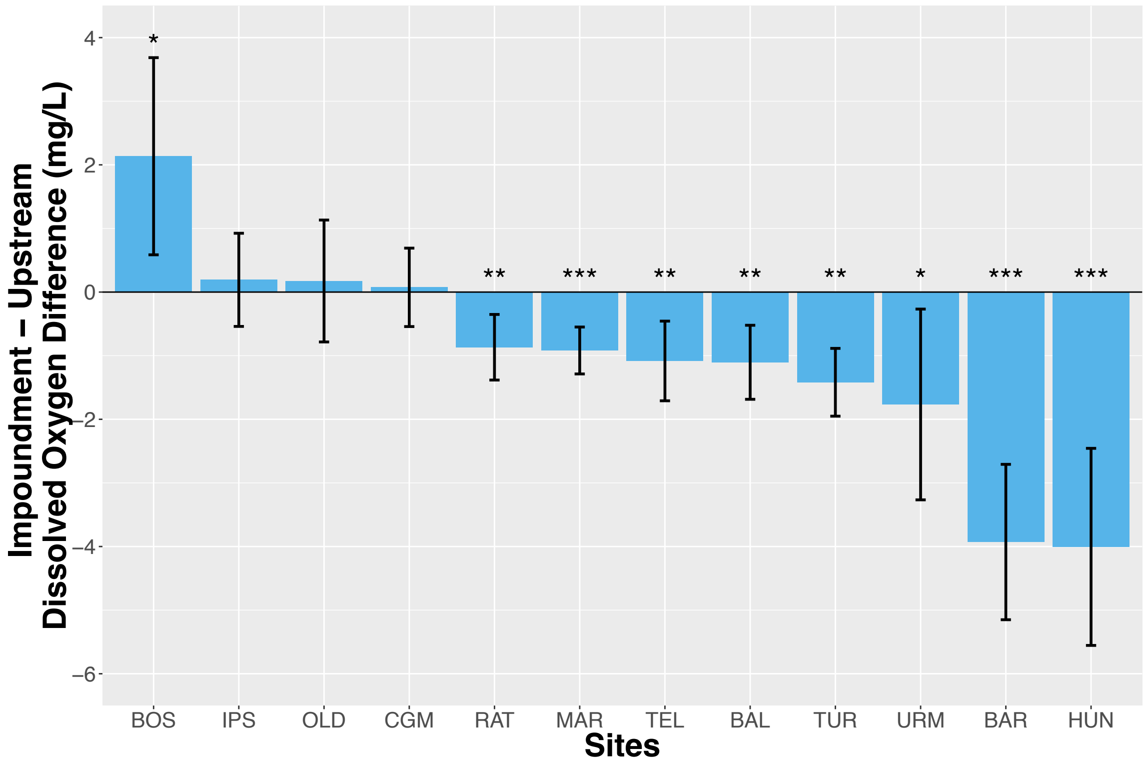

Figure 3.3 – Average impoundment DO difference (i.e., impoundment – upstream) at the 12 sites based on four, week-long deployments in summer 2015 and 2016 (n = 8 days for each site). Turner (TUR) data is from 3 week-long deployments in summer 2015 only (n = 6 days). See Table 3.1 for site abbreviations. Error bars represent the 95% confidence interval from a paired t-test (by date) comparing downstream and upstream temperatures. * $p< 0.05$, ** $p<0.01$, *** $p<0.001$.

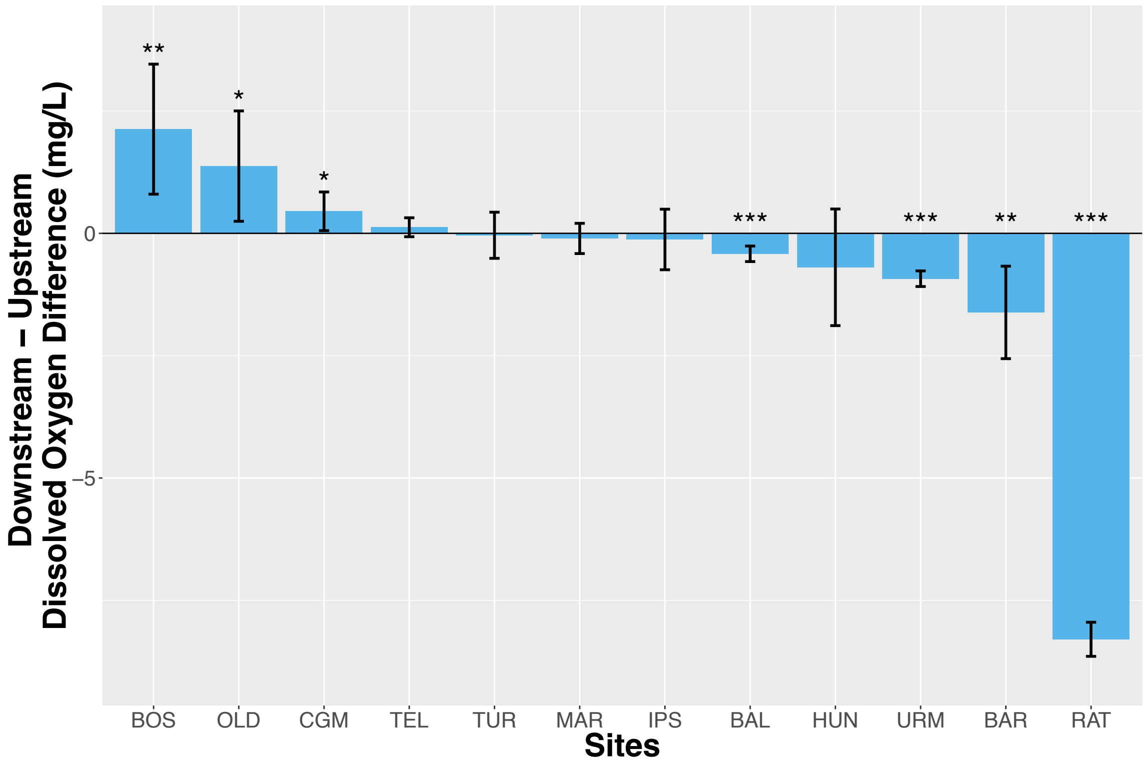

Figure 3.4 – Average downstream DO difference (i.e., downstream – upstream) at the 12 sites based on four, week-long deployments in summer 2015 and 2016 (n = 8 days for each site). Turner (TUR) data is from 3 week-long deployments in summer 2015 only (n = 6 days). See Table 3.1 for site abbreviations. Error bars represent the 95% confidence interval from a paired t-test (by date) comparing downstream and upstream temperatures. * $p< 0.05$, ** $p<0.01$, *** $p<0.001$.

CHAPTER 4

CONCLUSION

This thesis expands upon a limited body of literature detailing the impacts of small, surface-release dams to stream temperature and dissolved oxygen (DO) concentrations. The few studies previously published considered a small number of dam sites per study, and reported highly varying responses across studies, making it difficult to predict the impact of a small dam. This study investigated the impact of 30 dams to stream temperature and the impact of 12 dams to DO regimes. Such a large number of sites is unprecedented in the dam literature and allowed me to identify with more certainty the range of responses to dams.

The large number of sites shed light on a number of responses to small dams, both those that were observed consistently across sites and those that showed no consistent response. Impoundments were the most consistently and negatively impacted, and experienced the highest temperatures and generally the lowest DO concentrations within a reach. These warmed impoundment surface waters translated to warmer downstream temperatures at most sites (11 of 18), although there was a large range (0.20 – 5.25 °C) in the magnitude of warming. Five sites had no difference between upstream and downstream temperatures and two dams had cooler downstream temperatures. Whereas there was a pattern of impoundment warming leading to downstream warming, the general trend of impoundment DO losses did not carry as consistently downstream of the dam. As such, there was roughly an equal number of sites that experienced increases in downstream DO, no change in downstream DO, and decreases in downstream DO relative to their upstream reach.

These changes carry potential biological impacts as warmer temperatures and decreased DO can lead to elevated stress or death within ectothermic aquatic organisms and alter fundamental aspects of the stream ecosystem. At seven of the temperature sites, warming downstream of the dam was significant enough to shift an upstream cold- or cool-water thermal classification into a warmer one. The classifications referenced here were determined from temperatures at which significant changes in stream fish community occur in Connecticut and represent sites that likely are having direct impacts to biotic communities. Despite cooling rates, the warm temperatures persisted for an average of 1.7 km downstream of the dam, and suggest that the potential alterations to biotic communities may not be immediately recovered and extending several kilometers downstream. Three sites decreased DO downstream of the dam below a state threshold (5.0 mg/L) that may result in elevated stress for aquatic organisms. Four sites experienced hypoxia or anoxia within the impoundment that may carry more severe impacts to biota, in the form of being lethal to organisms and in the form of serving as hotspots for the methylation of mercury.

While these impacts carry biotic implications, not all sites were impacted the same way, and by quantifying variation in water quality responses across a large number of sites I could determine features of the dams and landscape that influence differences in response across sites where other studies did not have the sample size to do so. Broadly, the dams with the largest, most negative impacts (e.g., greatest temperature increases and DO decreases) were those with impoundments that represented the greatest change in physical characteristics relative to the upstream, un-impacted stream condition. Dam impoundments are flat-water, lentic environments, and those dams that generally imparted the greatest impact were those that created such a habitat on streams with the most contrasting conditions (e.g., high-gradient streams in small, highly-forested watersheds) upstream of the dam. Additionally, these impacts were most pronounced during periods of lowest flow, when the largest relative damming effect was achieved on stream flow.

Understanding the factors driving the severity of water quality impacts from dams will allow managers to better predict how temperature and DO are being impacted by dam sites across the landscape. This will in turn make it easier to identify sites within an agency's jurisdiction that are likely being most negatively impacted by a small dam. Those sites with the largest and most negative impacts from a small dam will likely experience the largest and most positive impacts from dam removal. Establishing more realistic and accurate expectations may be particularly useful in drawing in potential project partners looking to invest funding into a project most likely to return the largest benefits to water quality.

Data collected as a part of this thesis lays the groundwork (i.e., provides "before removal" data) for one of the largest to-date studies on the impacts of dam removal on water quality. As to be expected given the limited literature on the impacts of dams on water quality, studies investigating the impacts of dam removal to water quality are also limited and highly variable across the literature (Bushaw-Newton et al. 2002, Orr et al. 2006, Tuckerman and Zawiski 2007, Muehlbauer et al. 2009, Kornis et al. 2014, Bellmore et al. 2017). To have a benefit to water quality from dam removal, a site must first be negatively impacted by a dam, and sites with the largest negative impacts (e.g., most elevated impoundment and downstream temperatures, largest decrease in impoundment and downstream DO) should experience the greatest benefits following dam removal. The high-resolution spatial and temporal temperature and DO pre-dam removal data collected within this study will allow for robust comparisons of the effects of dam removal on temperature and DO in the future.

These results also provide insights into monitoring strategies to identify the impacts of small dams to water quality. Dams exhibited the largest downstream impacts during warm, summer months (June – September) and periods of lowest flow. Thus, monitoring efforts in New England should be focused during these warm periods (e.g., those with the largest impacts) to identify the ‘worst case’ effect of a dam. When planning for supplies and time investment, high-resolution data loggers should be used to collect continuous water quality data. The high-resolution loggers are continuously recording and can provide a more complete picture of conditions when a single, spot measurement does not capture diel variation, can record the most complete impact of the dam, and allow for the collection of such data at a large number of sites (e.g., the 160 locations with a temperature logger and the 36 locations with a DO logger deployed in this study) that would otherwise be impossible with spot-measurements made by personnel alone. At a minimum, loggers should be placed upstream of the impoundment, within the impoundment, and downstream of the dam. Temperature loggers are cheaper than DO loggers and installing 3-4 loggers downstream of the dam until a thermal barrier (e.g., tributary, estuary, impoundment) is reached will be able to capture both the thermal impact and thermal recovery from the dam. Installing an upstream logger was not feasible at several of my temperature sites due to access and the fact that several tributaries entered the impoundment. Without the upstream logger, the reference or undammed condition was not known, and prevented the calculation of downstream warming magnitude and dam thermal footprint. Additionally, without an upstream logger there is no ability to establish a target recovery temperature following restoration. Finally, while I had several years of data, the patterns and magnitudes of impact were similar across periods of summer base flow at these sites and suggest that the impact of a dam can be obtained via shorter-duration (e.g., single week- or month-long) deployments of loggers. Small dams are ubiquitous features of the landscape, and their impacts to water quality and stream ecosystems have been understudied and largely either over-simplified (e.g., assume they all have downstream warming) or overlooked (e.g., assume their impacts are not large enough to matter). As the most comprehensive examination of impacts of dams on temperature and DO, this thesis offers clarity into a muddled pool of results, and highlights that not all dams impact water quality in the same way. While some dams have little impact, others can have large and long-lasting impacts and should be the focus of dam removal efforts to restore stream ecosystem resiliency across the landscape.

APPENDICES

APPENDIX A

STANDARD OPERATING PROCEDURE FOR CONTINUOUS TEMPERATURE MONITORING

Prepared by: ______________________________
Peter Zaidel Date:

Reviewed by: ______________________________
Dr. Allison Roy Date:

____
Beth Lambert Date:

1.0 Scope and application

1.1 The purpose of this Standard Operating Procedure (SOP) is to detail standard methods for monitoring continuous temperature in wadeable streams and non-wadeable lakes and reservoirs. Following the steps and methods described in this document will ensure consistent deployment and retrieval of the data loggers, accurate readings by the logger, and a standardized management system of collected temperature data. This document references both the HOBO® Water Temp Pro V2 (U22-001) Manual and the HOBOware® User's Guide for specific steps in operating Onset© HOBO® Water Temp Pro V2 (U22-001) Data Loggers (referred to as 'HOBOs', 'data loggers', or 'loggers' in this SOP).

1.2 A properly calibrated and installed continuous data logger makes possible the collection of long-term environmental data at a resolution and scale that would not be practical or feasible without the use of the unattended and automatically recording logger. While this SOP details the use of Onset© HOBO® Water Temp Pro V2 (U22-001) Data Loggers, many other such loggers are available for water temperature monitoring, and with the appropriate changes (e.g., different user manual, different sizes for logger housing, etc.), the general process described in this document may be applicable to any logger used. Water temperature loggers can be deployed in fresh, brackish or saltwater environments for several years at a time, though more frequent site visits to ensure constant submersion, minimal biofouling, and downloading of data should be performed when possible.

2.0 Method summary

HOBO® data loggers should be installed in reaches of a stream with well-mixed water temperature. Before being deployed, loggers should pass a quality assurance (QA) ice bath check in the laboratory. In the field, a National Institute of Standards and Technology (NIST)-certified thermometer should be used to record discrete temperature measurements to determine representative locations in which to deploy the data loggers. The loggers should be secured within a white PVC flow-through housing unit. The housing unit and logger should be secured (e.g., via attachment to a tree, root, anchor, rebar) to maintain the desired sampling location. The temperature reading from the NIST-certified thermometer should be recorded during deployment and subsequent site visits to compare with the data logger as an in-field QA check. Although the loggers can be set to record as often as one reading per second, a larger interval, such as a reading every 15 minutes, achieves a similar level of precision in the data while keeping the data volume and file size significantly lower. Refer to Table 1 to determine the most project-appropriate logging interval.

3.0 Definitions and abbreviations

.csv	Comma separated value. A transitional file format that '.hobo' files can be exported out of HOBOware as to be used by statistical computing software such as Microsoft Excel or Program R
GPS	Global positioning system. Coordinates obtained through GPS are used to mark locations of data loggers on the Earth's surface
.hobo	File extension for data files created by the HOBO U22 data logger
Hyporheic zone	Area beneath and on either side of a streambed where groundwater and surface water mix.
MRPD	Mean residual pool depth. Provides a measure of pool depth that is not impacted by current flow conditions (i.e. depth at 0 flow)
PVC	Polyvinyl chloride. Used to make pipes (or lengths of pipes here) able to withstand long periods underwater without deteriorating
QA	Quality assurance. The steps taken to ensure that accurate data are collected
USGS	United States Geological Survey

4.0 Healthy and safety

The biggest safety concern with temperature monitoring is working in water. Large rivers and high velocities can increase safety risk in streams; while in lakes and reservoirs, the concern is about working over deep water, particularly during cold weather conditions. When working in reservoirs above low head dams, there is extreme risk of going over the dam and getting caught in the immediately downstream recirculating currents. Great care should be taken while in the field with crewmember safety being the top priority during these high-risk conditions. All crewmembers should have received and passed USGS "Over the Water Training", which specifically addresses issues related to working in streams and reservoirs behind low head dams. All crewmembers will wear Coast Guard approved life jackets, and properly fitting waders with the appropriate footwear attached. When the use of a canoe is required above a dam, an anchor will be deployed to hold the craft a safe distance away from the spillway. When working over the water, a two-person crew is required.

Figure A.1. Some dams in Massachusetts feature warning signs to alert those upstream of the dangers ahead. Many do not, however.

5.0 Personnel qualifications

The crew leader should have sufficient experience to direct the day-to-day activities in the field in a safe and efficient manner. The crew leader must also be experienced enough to provide guidance and instruction for less experienced assistants.

6.0 Equipment and supplies

6.1 Onset© data logger equipment and software

- Onset© HOBO® Water Temp Pro V2 (U22-001) Data Loggers

- Onset© HOBOware Pro software
- Onset© HOBO Optic USB Base Station
- Onset© HOBO U-DTW-01 Waterproof Shuttle

6.2 Ice bath QA

- Cooler
- Water / ice
- NIST-certified thermometer

6.3 Protective housings

- 1½" PVC piping
- 1½" PVC DWV Hub Cap
- 1½" PVC DWV Female Adapter
- 1½" PVC DWV Male Cleanout Plug
- PVC primer and glue
- Metric ruler
- Sharpie©
- Drill
- ¼" drill bit

6.4 Stream installations

- Onset© HOBO® Water Temp Pro V2 (U22-001) Data Logger
- Built Housing (see Section 6.3 above)
- 1/8" Vinyl-coated wire rope
- 1/8" Wire rope clips
- Nut driver
- Wire cutters
- NIST-certified thermometer
- 100-meter tape
- Meter stick
- Concave spherical densiometer
- GPS
- Flagging
- Sharpie©
- Camera

6.5 Impoundment installations

- All items from Section 6.4 – Stream installations
- Anchor (cinder block)
- Float (toilet tank float with a screw eye attached)

7.0 Procedure

7.1 Launching the logger

Whether preparing to deploy the logger for an ice bath QA in the lab or logging in the field, the U22 logger needs to be launched via HOBOware. Follow the steps in the HOBOware® User's Guide to connect the U22-001 to the computer via the Optic USB Base Station. Launching the loggers for an ice bath requires different logger settings than launching a logger to be deployed in the field, and caution should be exercised not to confuse the two settings. In any launching instance, if a warning pops up about the logger having a bad battery, do not use that logger. Only loggers with a green battery symbol labeled "Good" should be used for QA checks or deployment in the field. All others should be replaced.

7.1.1 Launching for an ice bath QA

1. Set the logger to record at a sampling interval of 1 minute and to begin recording an hour from the current time (e.g., if launching at 9:00AM, set to begin recording at 10:00AM. Note that 10:00 will be the start of the bath, in this example). This hour should provide ample time for the logger to equilibrate to the temperature of the ice bath.
2. Launch the logger by pressing 'start'.

7.1.2 Launching for deployment in the field

1. Set the logger to an appropriate sampling interval and to begin recording on the morning of the anticipated deployment date. Reference Table 1 to determine the appropriate logging interval and selected parameters (temperature only vs. temperature and battery voltage). The anticipated length of time before a planned site visit or download date, resolution of data collected, and amount of data space should influence the selection of a logging interval. It should be noted that even when the logger is only recording temperature, any dip in battery voltage below 3.1V (a "bad battery" event according to Onset©) would be recorded in the data file.
2. Launch the logger by pressing 'start'.

Table A.1. Time to fill the 64K of memory on a U22 Pro V2 temperature data logger based on selected logging interval and sampling parameters. Adapted from HOBOware Pro 08/20/2015.

Logging Interval	**Temperature Only**	**Temperature + Battery Voltage**
1 min	30.2 days	18.1 days
5 min	150.9 days	90.6 days
10 min	301.9 days	181.1 days
15 min	1.2 years	271.7 days
30 min	2.5 years	1.5 years
1 hour	5.0 years	3.0 years
2 hours	9.9 years	6.0 years

7.2 *Ice bath QA*

Before loggers can be deployed in the field, they must first be able to pass an ice bath QA check in the lab. Using methods described by Dunham et al. (2005), fill a cooler with liquid water and ice to bring the temperature of the water within the cooler to 0.0 °C. Verify the water is at 0.0 °C by placing a NIST-certified thermometer in the ice bath, and help maintain the temperature by covering the cooler. Do not place any loggers in the bath until 0.0 °C has been reached.

1. Follow the launch procedure described in Section 7.1.1 and fully submerge the loggers in the ice bath.
2. Check the temperature (using a NIST-certified thermometer) at the start of the bath, at the middle (30 minutes from bath start), and at the end of the bath (1 hour from the start) to ensure it remains at 0.0 °C.
3. After an hour of recording, remove the loggers from the bath.

7.2.1 *Reading the loggers out in HOBOware Pro*

Refer to the HOBOware® User's Guide on how to attach the logger to the computer and read out the data from the logger. Be sure the file is labeled with some identifying information (e.g., logger serial number, QA, water body name, sampling location, dates in the field, etc.). Whether reading out the data from a QA or from a field deployment, the plotted data file should be exported out of HOBOware as a comma separated file (.csv) for use with later analysis software (Microsoft Excel, Program R).

7.2.2 *Analyzing the QA data in Microsoft Excel*

Use Microsoft Excel to analyze the data collected by the loggers during the ice bath QA. For each logger calibrated, the date/time and the corresponding temperature recording at that time is required.

1. Open a blank Microsoft Excel file.
2. Copy and paste from the individual logger .csv files the temperature readings recorded during the hour of the ice bath.
3. Determine if any measurements are greater than 0.2 °C (the acceptable accuracy range of a U22) away from the NIST-certified reading. Use the Conditional Formatting function to highlight all cells that are less than -0.2 °C and all those that are greater than 0.2 °C. Any highlighted cells indicate a failed QA check for that logger.
4. A second ice bath QA should be performed on any loggers that failed the first. Those that fail a second QA should not be used.

7.3 Building logger housings

To help protect the HOBO Pro V2 Temperature Data Loggers from damage (collision or abrasion via objects in stream, sediments) and potential temperature-skewing direct solar radiation while in the field, the data loggers should be placed in "flow-through" white PVC housings (Dunham et al. 2005). To assemble the housings:

1. Cut 1½" diameter white PVC tubes into 13cm segments.
2. Use PVC primer and glue to attach a 1½" female adapter to one end of the pipe and a 1½" hub cap to the other end. Note that different primers and glues dry at different rates and the time to sufficiently dry will be dependent on the products used.
3. With a ¼" drill bit, drill 2 holes near the center of the cap ¼"–½" apart to feed wire cabling through during deployment.
4. Using the same ¼" drill bit, drill holes in the housing to allow water to flow through the unit. Drill through both sides of the main body of the housing (not the attached cap/adapter) four times to create two rows of evenly spaced holes (Figure A.2), and drill 1 hole through the end of the extruding square on the male plug.
5. Screw the male plug into the female adapter to complete the unit.

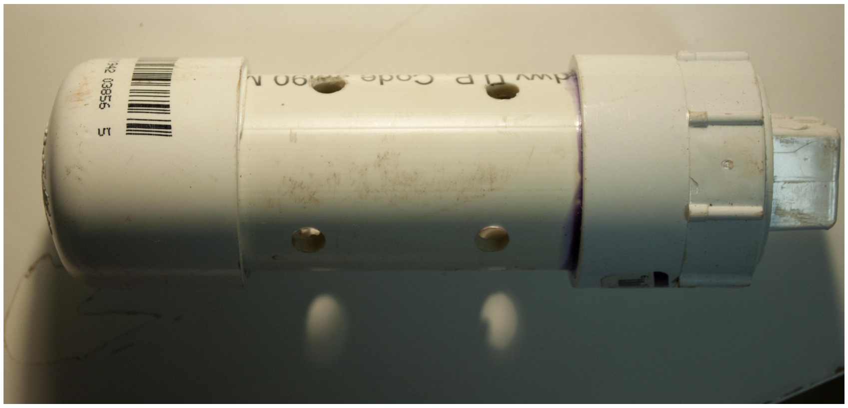

Figure A.2. A built logger housing showing the two rows of flow-through holes. The other four holes are located on the opposite side of the housing.

7.4 *Deploying the data loggers*

Before being deployed in the field loggers should be launched via HOBOware® Pro software following the steps described in Section 7.1.2. Select a location with temperatures representative of the reach of interest.

7.4.1 *Stream installations*

Probe various locations in the stream with a NIST-certified thermometer to identify zones of well-mixed and representative temperature in which to deploy the HOBO data loggers. Avoid installing loggers in low temperature (e.g., groundwater inflow) or high temperature (e.g., wastewater inflow) locations that are not representative of the stream. Care should be taken to thoroughly investigate the reach of interest to ensure that a representative sample location is chosen. Ensure the loggers have the best chance of staying submerged throughout summer low flows by choosing an installation location in a pool or a deep run within the stream. Note that using less cabling (i.e. a permanent structure closer to the stream bank) lowers the chances of the logger being stranded out of the water following a period of high flows.

1. Record the logger number on the data sheet.
2. Feed the vinyl-coated wire cable through one of the holes in the cap of the logger housing, through the eyehole at the top of the U22 logger, and back out the other hole in the cap.
3. Pull the logger into the housing so it sits snug at the top.
4. Slip a wire rope clip over both strands of wire cable as close to the top of the PVC housing as possible. Align the

rope clip so that the "loose" end of the cable is closest to the inside of the U. Use a nut driver to tighten the two hex nuts of the rope clip down on the wire rope to secure the logger in place at the top of the housing.

5. Use wire cutters to cut the length of cabling needed to reach from the selected in-stream deployment location to the permanent bank structure. Depending on the width of the permanent structure, enough cabling should be cut to be able to wrap around the structure and leave a few inches excess.
6. Wrap the cable around the permanent bank structure and use a nut driver to tighten the two hex nuts of the rope clip down on the two pieces of cable to secure the logger to the permanent bank structure. Note that the excess cabling is meant to act as "backup" in the off chance that high flows pull the housing downstream and some of the cabling through the wire rope clip.

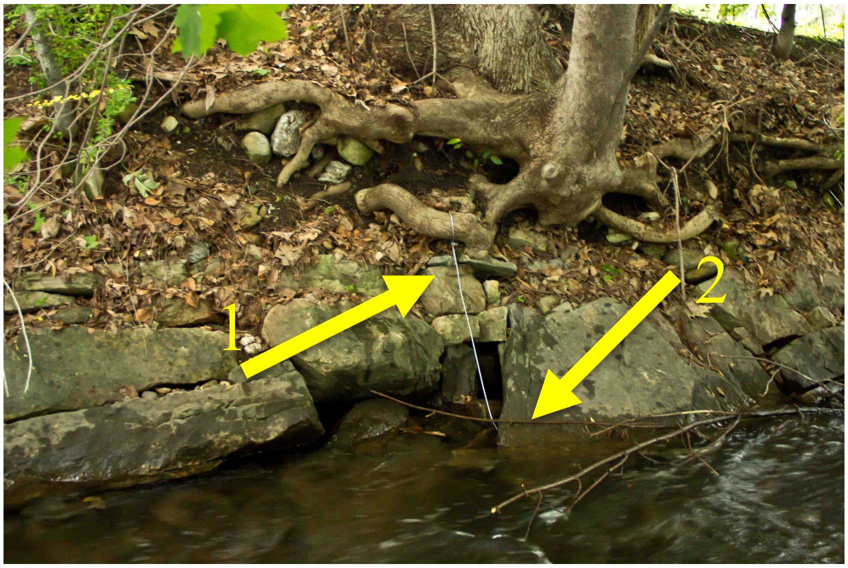

Figure A.3. Yellow arrow 1 indicates the permanent bank structure (root) being used to anchor the logger and housing in stream. Yellow arrow 2 indicates the large boulder and smaller rocks being used to protect and anchor the logger.

7. In high gradient streams, place rocks around and on top of the logger and the cabling to help anchor it, maintain its desired location during high flow events, and camouflage it from members of the public (Yellow arrow 2 in Figure

A.3). In silty, low gradient stream reaches (e.g., many coastal rivers), a heavy rock on top of the logger may sink it down into the hyporheic zone, and thus into cold groundwater inputs. In such an instance, the logger should be attached (e.g., zip ties, hose clamps) to a piece of rebar that has been hammered into the streambed (Figure A.4).

Figure A.4. A logger attached to a piece of rebar via two zip ties in a low gradient stream reach.

8. Record the time the logger was installed. Document the logger location by recording GPS coordinates, taking photos of the logger and stream, writing a brief site description, and sketching the stream reach (Attachment A.1).
9. Collect additional stream measurements that may influence temperature. Measure the depth of the water and the wetted width of the stream at the location of the logger. If the logger is deployed in a pool, measure the maximum pool depth and the depth at the outlet of the pool to determine the mean residual pool depth (MRPD). Subtract the outlet depth from the depth at the point of the logger and record this value as the MRPD. Measure tree canopy cover at the water surface above the logger with a spherical densiometer

in 4 directions – upstream, river right, downstream, and river left (Attachment A.1).

7.4.2 *Lake installations*

When installing U22 data loggers within a lake or reservoir, the desired goal of the research should dictate the installment location. If identifying surface water temperatures is the goal, then a single logger attached to a buoy should suffice. If determining if the lake is stratified, one logger near the surface as well as one near the bottom may be required to identify differences in temperature representative of a summer stratified lake environment (i.e. higher temperature at the surface and lower temperature near the bottom). In either of these instances, the logger(s) should hang under a float attached to an anchor on the lake bottom. If interested in transitional, littoral zones, the permanent bank cabling method for installation described in Section 7.4.1 may be used. Consideration of public visibility or ease of access for vandals should also be considered when determining the exact deployment location. As with the stream installations, a NIST-certified thermometer should be used to identify zones of well-mixed and representative temperatures. The steps below detail the installation of a logger suspended below a float and connected to an anchor.

1. Record the logger number on the data sheet.
2. Feed the vinyl-coated wire cable through one of the holes in the cap of the logger housing, through the eyehole at the top of the U22 logger, and back out the other hole in the cap.
3. Pull the logger into the housing so it sits snug at the top.
4. Leave roughly half a meter of extra cable to attach the housing to the float. Use wire cutters to cut the cable.
5. Slip a wire rope clip over both strands of wire cable to as close to the top of the PVC housing as possible. Use a nut driver to tighten the two hex nuts of the rope clip down on the wire rope to secure the logger in place at the top of the housing.
6. Feed the free end of the cable through the loop at the bottom of the float, pulling tightly to bring the housing right below the float. Secure in place with a wire rope clip.

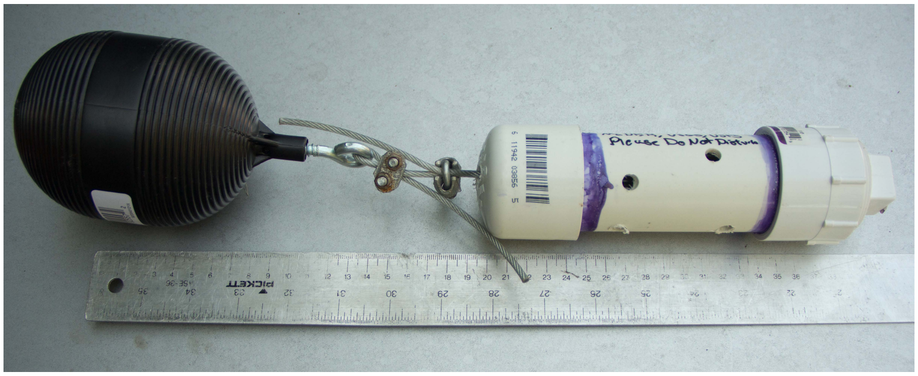

Figure A.5. A logger housing attached beneath a float.

7. Loop a vinyl-coated wire cable through the opening of a cinder block and use a wire rope clip to fasten the wire cable onto itself. Note that rope can be used in place of the wire cable for this.
8. If the depth is not known, lower the cinder block to determine the depth at that location. If access of vandals is not a concern, leave roughly half a meter of additional cable to fasten the cable line to the float while still leaving enough slack to allow the float to remain at the surface if the water levels rise. If vandalism is a concern, and doing so is in accordance with project goals, less line can be used to sink or "hide" the buoy just below the surface of the water.
9. Once the appropriate amount of cabling has been determined, use wire cutters to cut the rope. *NOTE: Be sure to include about 10-15cm of additional wire to be used to loop through the float and fasten the cable line to the buoy.*
10. Lower the cinder block to the desired location within the lake to deploy the logger.

Figure A.6. A logger hanging beneath a float in a lake.

11. Record the time the logger was installed. Document the logger location by recording GPS coordinates, taking photos of the logger and lake, writing a brief site description, and sketching the lake reach.
12. Collect additional stream measurements that may influence temperature. Measure the depth of the water and the canopy cover at the water surface above the logger with a spherical densiometer in 4 directions – upstream, river right, downstream, and river left (Attachment A.1).

7.5 *Lake vertical profile*

Due to the potential for stratification within the lake environment, a vertical profile of temperature and dissolved oxygen (if available) should be taken using a sonde. Use a marked line to accurately measure the depth of the lake. Regardless of depth, each profile should begin with a reading just below the surface of the water, and continue until 0.5 meters above the bottom of the lake. If the lake is less than 3 meters deep, following the surface sample, a reading should be taken every 0.5 meters until 0.5 meters from the bottom. If the lake is greater than 3 meters deep, a reading should be taken at an interval of 1 meter starting just below the surface and continuing until 0.5 meters above the bottom. The sonde should sit for three minutes at each depth to allow ample time for the probes to equilibrate. The sample values should be recorded on the data sheet (Attachment A.2).

7.6 *Mid-deployment checks*

Once deployed, loggers should be visited at least twice a year (spring and fall) to ensure they are still at the correct location and to download data. Before the logger is removed from the stream and the data downloaded, however, an in-field QA check should be performed on the logger. Place the probe of a NIST-certified thermometer next to the data logger and record the time of measurement and the recorded temperature for cross-referencing when the data is later downloaded onto the computer. Note the column labeled 'QA Temp' on the Continuous Temperature Deployment and Retrieval Field Data Sheet (Attachment A.1) with space for such measurements.

Once a year, loggers will be removed from the field and replaced with a logger that has passed the ice bath QA check to ensure a high level of confidence in accurate data collection. Loggers that are retrieved from the field must be calibrated with an ice bath (Section 7.2) before they can be re-deployed in the field. This trend of bi-annual logger checks and annual ice bath QA should continue through project completion.

7.7 Reading the loggers out in HOBOware Pro

See section 7.2.1 on how to read a logger out following retrieval from the field.

8.0 Data records and management

HOBO® U22 loggers record data and save it as a .hobo file when downloaded onto the computer. To be analyzed and used in other computing and analysis programs, the .hobo files should be exported from HOBOware as .csv files. To minimize the possibility of data loss, save all original .hobo and the corresponding .csv files on the computer, as well as duplicate copies of all files on a different computer or external hard drive. Though addressed in Section 7.2.1, it is imperative that all files are appropriately named with easily identifying information (e.g., the logger ID number, the stream name in which it was deployed, the dates of deployment, etc.).

9.0 Quality assurance procedures

All U22 data loggers should be calibrated via the ice bath method described in Section 7.2. At a minimum, in-field QA spot checks (Section 7.6) can be done bi-annually when the data is downloaded off of the logger, though additional QA temperature measurements may be taken when on site for other surveys throughout the year.

As a higher resolution field QC check, a minimum of 10% of data loggers should be duplicated via the installation of a second data logger alongside the logger being checked. This would allow not just a single point check, but rather a period of continuous checks. These duplicate data loggers should be deployed for approximately one week alongside the logger of interest before being moved to a different site. Given a logger accuracy of ±0.21 °C, an acceptable divergence between the two loggers would be 0.42 °C.

10.0 References

Dunham, J.B., Chandler, G.L., Rieman, B.E., Martin, D., 2005, Measuring stream temperature with digital data loggers- A user's guide: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Center General Technical Report RMRS-GTR-150WWW. http://www.fs.fed.us/rm/pubs/rmrs_gtr150.pdf.

Onset. 2012. HOBO® Water Temp Pro V2 (U22-001) Manual. Bourne, MA: Onset Computer Corporation <http://www.onsetcomp.com/files/manual_pdfs/10366-G-MAN-U22-001.pdf> Accessed 19 May 2015.

Onset. 2015. HOBOware® User's Guide. Bourne, MA: Onset Computer Corporation <http://www.onsetcomp.com/files/manual_pdfs/12730-S%20HOBOware%20User%27s%20Guide.pdf> Accessed 21 Aug 2015.

Attachment A.1

Continuous Temperature Deployment and Retrieval Data Sheet

CONTINUOUS TEMPERATURE DEPLOYMENT AND RETRIEVAL DATA SHEET

(FRONT)

Waterbody Name:	**Date**:
Crew:	
Address:	
Driving Directions/Parking:	
Logging Start Date / Time:	**Logging Interval**:

<table>
<tr><th>Sample Location Code</th><th>Lat / Long</th><th>Site Description</th><th>Stream Parameters</th><th>QA Temp (°C)</th><th>Densiometer</th><th>Photos</th></tr>
<tr><td></td><td></td><td></td><td>Logger ID #
Wetted width (m) =
Depth (cm) =
MRPD (cm) =
Time in =
Circle as applicable:
Lake / Stream
Riffle / Pool / Run
Streambed geology:</td><td><table><tr><th>Date</th><th>Time</th><th>Temp</th></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr></table></td><td>US:
RR:
DS:
RL:</td><td></td></tr>
<tr><td></td><td></td><td></td><td>Logger ID #
Wetted width (m) =
Depth (cm) =
MRPD (cm) =
Time in =
Circle as applicable:
Lake / Stream
Riffle / Pool / Run
Streambed geology:</td><td><table><tr><th>Date</th><th>Time</th><th>Temp</th></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr><tr><td></td><td></td><td></td></tr></table></td><td>US:
RR:
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<tr><td></td><td></td><td></td><td>Logger ID #
Wetted width (m) =
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Time in =
Circle as applicable:
Lake / Stream
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RR:
DS:
RL:</td><td></td></tr>
</table>

CONTINUOUS TEMPERATURE DEPLOYMENT AND RETRIEVAL DATA SHEET

(Back)

Site Sketch:

Attachment A.2

Lake Multiparameter Vertical Profile Data Sheet

LAKE MULTIPARAMETER VERTICAL PROFILE DATA SHEET

(Front)

Waterbody Name:	**Date**:	**Crew**:
Lake Depth:	**Sampling Interval (circle one)**	0.5m / 1.0m

GPS Lat:	**GPS Long**:
Site description:	

Depth (m)	**Recorded Parameters**			
	Temperature (°C)	**DO concentration (mg/L)**	**DO % Saturation**	**Conductivity (µs/cm)**
Surface (0.1 m)				

Notes:

LAKE MULTIPARAMETER VERTICAL PROFILE DATA SHEET

(Back)

Site Sketch:

APPENDIX B

STANDARD OPERATING PROCEDURE FOR CONTINUOUS DISSOLVED OXYGEN MONITORING

Prepared by: ______________________________
Peter Zaidel Date:

Reviewed by: ______________________________
Dr. Allison Roy Date:

____
Beth Lambert Date:

1.0 Scope and application

1.1 The purpose of this Standard Operating Procedure (SOP) is to describe standard methods for monitoring continuous dissolved oxygen (DO) levels in wadeable streams and non-wadeable lakes and reservoirs. Following the steps and methods described in this document will ensure consistent calibration, deployment and retrieval of the data loggers, and a standardized management system of collected dissolved oxygen data. This document references the HOBO® Dissolved Oxygen Logger (U26-001) Manual for specific steps in operating Onset© HOBO® U26-001 Dissolved Oxygen Data Loggers (referred to as ‘HOBOs’, ‘data loggers’ or ‘loggers’ in this SOP).

1.2 A properly calibrated and installed continuous data logger makes possible the collection of long-term environmental data at a resolution and scale that would not be practical or feasible without the use of the unattended and automatically recording logger. This SOP details the use of Onset© U26-001 Dissolved Oxygen Data Loggers to collect such fine-scale, continuous environmental data. However, many other such loggers are available for dissolved oxygen monitoring, and with the appropriate changes (e.g., different user manual, different sized logger housing, etc.), the general process described in this document may be applicable to any logger used. Dissolved oxygen loggers can be deployed in fresh, brackish or saltwater environments with any range of oxygen levels. When deployed in brackish or saltwater, conductivity measurements are required to adjust for changes in the saturation value of dissolved oxygen resulting from increased salt concentrations.

2.0 Method summary

HOBO® data loggers should be installed in reaches of a stream with well-mixed dissolved oxygen levels. Before deploying loggers, they should be calibrated in the laboratory. In the field, a separate probe should be used to record discrete dissolved oxygen measurements to determine representative locations of well-mixed oxygen in which to deploy the data loggers. The loggers should be secured within a white PVC flow-through housing unit. The housing unit and logger should be secured (via attachment to a tree, root, anchor, rebar) to maintain the desired sampling location. The multiparameter probe should log a discrete sample of dissolved oxygen and temperature at the deployment and retrieval of the logger to compare with the U26 and help account for potential drift due to biofouling. Although the loggers can be set to record as often as 1 reading per minute, a larger interval, such as a reading every 15 minutes, achieves a similar level of precision in the data while keeping the data volume and file size significantly lower.

3.0 Definitions and abbreviations

.csv	Comma separated value. A transitional file format that '.hobo' files can be exported out of HOBOware as to be used by statistical computing software such as Microsoft Excel or Program R
D.O.	Dissolved oxygen
GPS	Global positioning system. Coordinates obtained through GPS are used to mark locations of data loggers on the Earth's surface
.hobo	File extension for data files created by the HOBO U26 data logger
MRPD	Mean residual pool depth. Provides a measure of pool depth that is not impacted by current flow conditions (i.e. depth at 0 flow)
MSDS	Material safety data sheet. Provides a list of the hazards of a particular chemical, how to handle that chemical, and what to do in case of an accident
NWS	National Weather Service
PVC	Polyvinyl chloride. Used to make pipes (or lengths of pipes here) able to withstand long periods underwater without deteriorating
USGS	United States Geological Survey

4.0 Healthy and safety

The biggest safety concern with dissolved oxygen monitoring is working in water. Large rivers and high velocities can increase safety risk in streams; while in lakes and reservoirs, the concern is about working over deep water, particularly during cold weather conditions. When working in reservoirs above low head dams, there is extreme risk of going over the dam and getting caught in the immediately downstream recirculating currents. Great care should be taken while in the field with crewmember safety being the top priority during these high-risk conditions.

All crewmembers should have received and passed USGS "Over the Water Training", which specifically addresses issues related to working in streams and reservoirs behind low head dams. All crewmembers will wear Coast Guard approved life jackets, and properly fitting waders with the appropriate footwear attached. When the use of a canoe is required above a dam, an anchor will be deployed to hold the craft a safe distance away from the spillway. When working over the water, a two-person crew is required.

Figure B.1. Some dams in Massachusetts feature warning signs to alert those upstream of the dangers ahead. Many do not, however.

5.0 Personnel qualifications

The crew leader should have sufficient experience to direct the day-to-day activities in the field in a safe and efficient manner. The crew lead must also be experienced enough to provide guidance and instruction for less experienced assistants.

6.0 Equipment and supplies

6.1 Onset© data logger equipment and software

- Onset© HOBO U26-001 Dissolved Oxygen Data Loggers
- Onset© HOBOware® Pro software (U26 *cannot* communicate with the basic HOBOware® software package)
- Onset© HOBO Optic USB Base Station

- Onset© HOBO U-DTW-01 Waterproof Shuttle
- Onset© U26 RDOB-01 Replacement DO Sensor Cap

6.2 Calibration

- Rubber calibration boot
- Freshwater-dampened sponge
- Sodium sulfite solution
- Small beaker

6.3 Protective housings

- 2” PVC piping
- 2” PVC DWV Hub Cap
- 2” PVC DWV Female Adapter
- 2” PVC DWV Male Cleanout Plug
- PVC primer and glue
- Metric ruler
- Sharpie©
- Drill
- ¼” drill bit

6.4 Stream installation

- Onset© U26-001 Dissolved Oxygen Data Logger
- Built Housing (Section 6.3)
- Rebar
- Hammer
- 1/8” Vinyl-coated wire rope
- 1/8” Wire rope clips
- Nut driver
- Wire cutters
- 8” zip ties
- 3” Hose clamps
- 100-meter tape
- Meter stick
- Concave spherical densiometer
- GPS
- Flagging
- Sharpie©
- Camera

6.5 Lake installation

- Onset© U26-001 Dissolved Oxygen Data Logger

- Built Housing (Section 6.3)
- Anchor (cinder block)
- Float (buoy)
- 1/8” Vinyl-coated wire rope
- 1/8” Wire rope clips
- Nut driver
- Wire cutters
- 100-meter tape
- Concave spherical densiometer
- GPS
- Camera

7.0 Procedure

7.1 Calibration

1. Each U26 logger requires a working sensor cap. Sensor caps last for six months and once expired, the logger will not function. Regardless of use, the caps have a shelf life, or expiration date, after which time the cap is no longer usable. The expiration date can be found on the label of the canister each cap is packaged in. To install a new cap on the probe, follow the steps outlined in the HOBO® Dissolved Oxygen Logger (U26-001) Manual.
2. Perform a two-point calibration to both 100% and 0% oxygen levels. Refer to the methods described in the HOBO® Dissolved Oxygen Logger (U26-001) Manual to calibrate the logger. *NOTE: Should the Optic USB Base Station be disconnected from the logger’s optic port during calibration, the procedure will need to be restarted. Care should be taken to avoid loss of connection between the two devices.*
3. As is stated in the Manual, the current barometric pressure at the time of the calibration is required, and this value can be obtained in one of several ways. It is preferred that a discrete reading be obtained during the calibration with either a multiparameter probe or a barometer. If this is not possible, barometric pressure data is available from the National Weather Service (NWS). NWS normalizes these data to sea level, and if using this value, the Lab Calibration tool in HOBOware Pro will also require the current elevation to convert the normalized sea level value.
4. Ensure that the water dampened sponge (for 100% saturation) and the sodium sulfite solution (for 0% saturation) are at room temperature to minimize the equilibration time required by the logger to the temperature of either solution.

5. After performing both the 100% and 0% calibrations, record the concentration values from the logger, the gain, and the offset for future reference (Attachment B.1).

7.2 Launching the loggers

Once the logger has been calibrated, it may be launched via HOBOware Pro. Ensure the battery condition is marked as "Good", and do not use any logger with a bad battery. Note however that slight clockwise or counterclockwise rotations of the logger within the coupler may properly align the optic ports of the logger and base station and remedy a "bad battery" notice. To maintain a high resolution of data, yet also keep data volume at a reasonable level, set the logger to record every 15 minutes. The logger needs to be logging when the Reference Calibration (Section 7.4.3) is performed at installation. To ensure this, set the logger to begin recording data at 6:00AM on the morning of the planned deployment. Name the file with an easily identifying name (e.g., the logger ID number, the stream name in which it will be deployed, etc.).

7.3 Building logger housings

To protect the HOBO from damage in the stream and from exposure to temperature-altering direct solar radiation, the loggers will sit in "flow-through" white PVC housings. To assemble the housings:

6. Cut 2" diameter white PVC tubes into 33cm segments.
7. Use PVC primer and glue to attach a 2" female adapter to one end of the pipe and a 2" hubcap to the other end.
8. With a ¼" drill bit, drill 2 holes near the center of the cap ¼"–½" apart to feed the wire cabling through.
9. Using the same ¼" drill bit, drill holes in the housing to allow water to flow through the unit and to feed the securing zip ties through. Drill through both sides of the main body of the housing (no attached cap/adapter) six times to create three rows of 12 evenly spaced holes (Figure 2). Drill 1 hole through the end of the extruding square on the male plug.
10. Screw the male plug into the female adapter to complete the unit.

Figure B.2. An assembled logger housing showing the three rows of flow-through holes. The other six holes are on the opposite side of the housing.

7.4 *Deploying the data loggers*

Before being deployed in the field, loggers should be launched via HOBOware® Pro software following the steps described in Section 7.2. Select a location with dissolved oxygen levels representative of the entire reach. To determine a representative location, use a calibrated sonde (Appendix B) that can take and display instantaneous dissolved oxygen readings. D.O. meters and loggers generally take several minutes (e.g., 15 minutes for HOBO U26, 5 minutes for Hanna Instruments 9829 multiparameter sonde) to equilibrate to stream conditions and accurately measure dissolved oxygen. Details on time to reach equilibrium can be found in the device manual. Once equilibrated, probe various locations throughout the stream to determine the oxygen levels of the stream and a representative location. Allow a minute for the sonde to stabilize its readings before recording when moving throughout the water body.

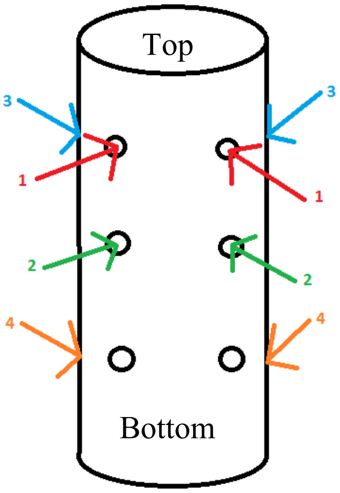


Figure B.3. Identification of zip tie attachment locations referenced in Sections 7.4.1 and 7.4.2.

7.4.1 *Stream Installations*

Probe various locations in the stream with a pre-calibrated sonde to identify zones of well-mixed and representative oxygen levels in which to deploy the HOBO data loggers. Avoid installing loggers in non-representatively low (e.g., too close to shore, too deep, site of high decomposition) or high (e.g., immediately below a dam spillway, in a riffle, in a macrophyte bed) locations within the stream. Care should be taken to thoroughly investigate the reach of interest to ensure that a representative sample location is chosen. Determine a representative reach of stream in which to deploy the loggers, and locate a zone of streambed into which several feet of rebar can be hammered in (i.e. avoid bedrock).

1. Record the logger number on the data sheet.
2. Feed an 8” zip tie through 2 adjacent holes in the logger housing (Red arrows labeled #1 in Figure B.3).
3. Push the zip tie so that it is flush in a U-shape across the inside of the opposite wall of the PVC.

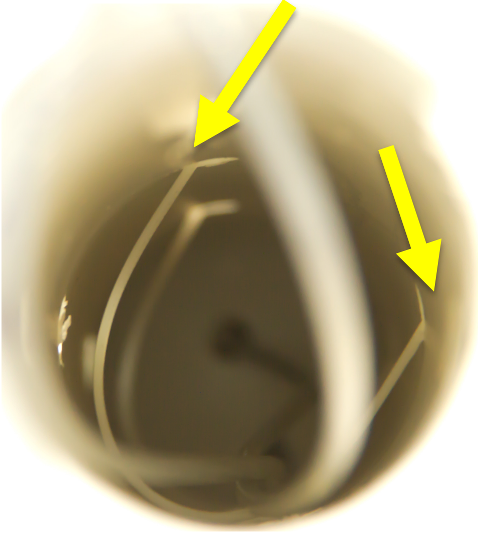


Figure B.4. A zip tie flushed in a U-shape across the inside of the PVC (arrows point to the holes in the housing that the zip tie was fed through).

4. Repeat steps 4&5 with another zip tie along the same side as the previously inserted zip tie (Green arrows labeled #2 in Figure 3).
5. Feed the vinyl-coated wire rope through one of the holes in the cap of the housing, down through the housing and the two U’s formed by the zip ties, through the eyehole at the top of the U26 logger, and back (still through the zip tie U’s) out the other hole in the cap.

6. Pull the logger into the housing so it sits snug at the top.
7. Slip a wire rope clip over both strands of wire cable to as close to the top of the PVC housing as possible. Use the nut driver to tighten the two hex nuts of the rope clip down on the wire rope to secure the logger in place at the top of the housing.
8. Fasten both zip ties to prevent lateral movement of the logger within the housing unit during deployment.
9. Hammer a 4' piece of rebar into the stream bed to a depth that ensures it is secure and will not be washed away under high flows. Note that in instances where rebar cannot be easily hammered into the streambed (e.g., bedrock), the loggers can be cabled to a permanent bank structure as detailed in Section 7.4.1 of the accompanying Continuous Temperature Monitoring Standard Operating Procedure.
10. Use two zip ties and two hose clamps to tighten the logger housing to the rebar. Zip ties should be fed through the opposite side as those installed in steps 2-4 and through the top most and bottom most rows of holes (Blue and orange arrows labeled #3 & #4 in Figure B.3). Hose clamps should also be secured near the top and bottom of the logger over the lengths of pipe that sit just inside the female adapter and the hubcap (Yellow arrows in Figure B.5).
11. Record the time the logger was installed. Document the logger location by recording GPS coordinates, taking photos of the logger and stream, writing a brief site description, and sketching the stream reach (Attachment B.3).
12. Collect additional stream measurements that may influence temperature. Measure the depth of the water and the wetted width of the stream at the location of the logger. If the logger is deployed in a pool, measure the maximum pool depth and the depth at the outlet of the pool to determine the mean residual pool depth (MRPD). Subtract the outlet depth from the depth at the point of the logger and record this value as the MRPD. Measure tree canopy cover at the water surface above the logger with a spherical densiometer in 4 directions – upstream, river right, downstream, and river left (Attachment B.3).

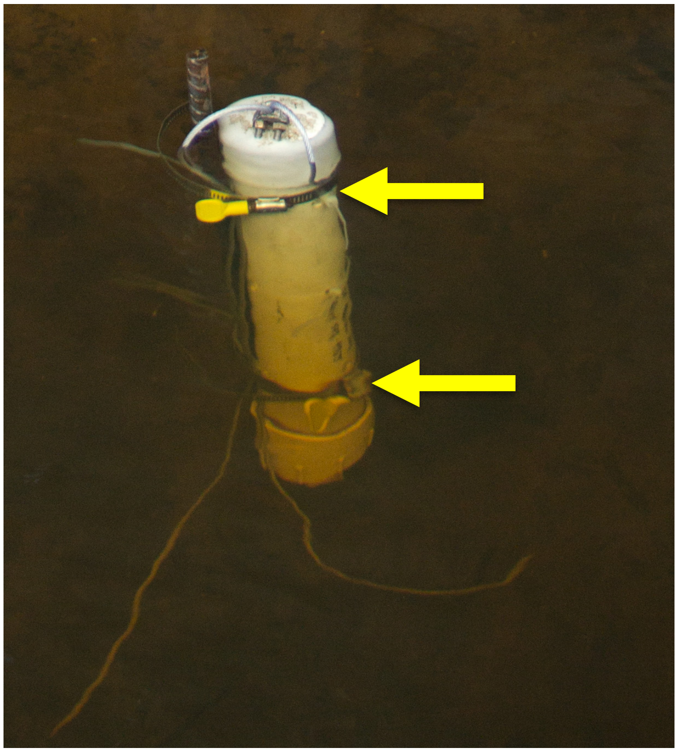


Figure B.5. A dissolved oxygen logger installed on a piece of rebar. Note that the protrusion of the rebar and housing out of the water is permissible if conditions (e.g.,, streambed geology or water depth) do not allow for full submersion. The yellow arrows indicate the location of the hose clamps.

7.4.2 *Lake installations*

When installing U26 data loggers within a lake or reservoir, the desired goal of the research should dictate the installment location. If identifying surface dissolved oxygen levels are the goal, then a single logger attached beneath a buoy should suffice. If determining if the lake is stratified, one logger near the surface as well as one near the bottom may be required to identify differences in dissolved oxygen representative of a stratified lake environment (i.e. high oxygen at the surface, low oxygen near the bottom). In either of these instances, the logger(s) should hang under a buoy attached to an anchor on the lake bottom. If interested in transitional, littoral zones, the rebar method for installation described in Section 7.4.1 may be used. As with the stream installations, a multiparameter sonde should be used to identify zones of well mixed and representative oxygen levels. Due to the depth of lake sites and the potential for resulting stratification, a vertical profile should be created of the lake using the multiparameter sonde (Section 7.5). The steps below detail the installation of a floating array connected to an anchor.

1. Record the logger number on the data sheet.
2. Feed an 8” zip tie through 2 adjacent holes in the logger housing (Red arrows labeled #1 in Figure B.3).
3. Push the zip tie so that it is flush in a U-shape across the inside of the opposite wall of the PVC (Figure B.4).
4. Repeat steps 4&5 with another zip tie along the same side as the previously inserted zip tie (Green arrows labeled #2 in Figure B.3).
5. Feed the vinyl-coated wire rope through one of the holes in the cap of the housing, down through the housing and the two U’s formed by the zip ties, through the eyehole at the top of the U26 logger, and back (still through the zip tie U’s) out the other hole in the cap.
6. Pull the logger into the housing so it sits snug at the top.
7. Slip a wire rope clip over both strands of wire cable to as close to the top of the PVC housing as possible. Use the nut driver to tighten the two hex nuts of the rope clip down on the wire rope to secure the logger in place at the top of the housing.
8. Fasten both zip ties to prevent lateral movement of the logger within the housing unit during deployment.
9. Feed some of the excess cable that was used to secure the logger within the housing through the loop at the bottom of the buoy, and use a wire rope clip to fasten the cable to itself, thus securing the logger in place below the buoy.

Figure B.6. Built logger housing attached to a buoy.

10. Loop a vinyl-coated wire cable through the opening of a cinder block and use a wire rope clip to fasten the wire cable onto itself. Note that rope can be used in place of the wire for this.
11. If the depth is not known, lower the cinder block to determine the depth at that location. If access of vandals is not a concern, leave roughly half a meter of additional cable to fasten the cable line to the float while still leaving enough slack to allow the float to remain at the surface if the water levels rise. If vandalism is a concern, and doing so is in accordance with project goals, less line can be used to sink or "hide" the buoy below the surface of the water.
12. Once the appropriate amount of cabling has been determined, use wire cutters to cut the rope. *NOTE: Be sure to include about 10-15cm of additional wire to be used to loop through the float and fasten the cable line to the buoy.*
13. Lower the cinder block to the desired location within the lake to deploy the logger.
14. Record the time the logger was installed. Document the logger location by recording GPS coordinates, taking photos of the logger and stream, writing a brief site description, and sketching the stream reach (Attachment B.3).
15. Collect additional measurements that may influence temperature. Measure the depth of the water at the location of

the logger. Measure canopy cover at the water surface above the logger with a spherical densiometer in 4 directions – upstream, river right, downstream, and river left (Attachment B.3).

7.4.3 *Reference calibration*

Once the deployment site has been determined, both for stream and lake sites, an official reference field calibration should be performed to verify the logger's readings. This same procedure should be performed when the logger is retrieved from the field to account for any potential drift in data as a result of biofouling.

1. Place a sonde in the water next to the data logger (Figure 7). *NOTE: The U26 data logger should be recording data at this point as this calibration will be used against the values obtained from the data logger upon download.*
2. To accurately measure water parameters, both units (logger and sonde) must equilibrate to water temperature. HOBOs require 15 minutes to equilibrate to water temperature, while most multiparameter sondes require less (verify with the appropriate user manual). Placing both units (HOBO and sonde) in the water while installing can reduce wait time for achievement of equilibrium.
3. Upon reaching equilibrium, record the date, time, DO concentration and percent saturation, temperature, and conductivity on the field data sheet (Attachment B.3).

Figure B.7. Performing a reference calibration with the sonde placed immediately next to the installed dissolved oxygen logger.

7.5 Lake vertical profile

Due to the potential for stratification within the lake environment, a vertical profile of temperature, dissolved oxygen, and conductivity should be taken using a sonde. Use a marked line to accurately measure the depth of the lake. Regardless of depth, each profile should begin with a reading just below the surface of the water, and continue until 0.5 meters above the bottom of the lake. If the lake is less than 3 meters deep, following the subsurface sample, a reading should be taken every 0.5 meters until 0.5 meters from the bottom. If the lake is greater than 3 meters deep, a reading should be taken at an interval of 1 meter starting just below the surface and continuing until 0.5 meters above the bottom. The values should be recorded on the data sheet (Attachment B.4).

7.6 Retrieving the data loggers from the field

Before removing the logger from the site, a retrieval calibration should be performed with a multiparameter probe. This will be used to help correct for any potential drift in dissolved oxygen readings as a result of biofouling during the logger's deployment. Follow the steps listed in Section 7.4.3. Only the sonde will need to equilibrate, and the respective user manual should be consulted to determine the length of time required for equilibration by the unit.

Once the calibration values have been recorded, remove all installation equipment (e.g., logger housing, rebar, cabling, cinder block, float) from the field. Data from the loggers should be downloaded (Section 7.7) and the installation equipment cleaned before being installed in a different water body.

7.7 Reading the loggers out in HOBOware Pro

Follow the steps outlined in the HOBO® Dissolved Oxygen Logger (U26-001) Manual to connect the logger to the computer, readout the data from the HOBO, and download it onto the computer. The files should be exported from HOBOware Pro as .csv files for use in later analysis in other computing programs.

8.0 Data records and management

HOBO® U26 loggers record data and save it as a .hobo file when downloaded onto the computer. To be analyzed and used in other computing and analysis programs, the .hobo files should be exported from HOBOware Pro as .csv files. To minimize the possibility of data loss, save all original .hobo and the corresponding .csv files on the computer, as well as duplicate copies of all files on a different computer or external hard drive. Though addressed in Section 7.2, it is imperative that all files are appropriately named with easily identifying information (e.g., the logger ID number, the stream name in which it was deployed, the dates of deployment, etc.).

9.0 Quality assurance procedures

A two-point calibration to 100% and 0% oxygen, as described in Section 7.1, should be performed on each U26 data logger to ensure the most accurate data collection. To account for any potential drift in biofouling, as described in Section 7.4.3, record the dissolved oxygen (both concentration and percent saturation), conductivity, and temperature with a handheld sonde at the deployment and retrieval of the logger. To correct for biofouling drift or varying conductivity in estuarine environments, follow the steps outlined in the HOBO® Dissolved Oxygen Logger (U26-001) Manual for using the HOBOware® Dissolved Oxygen Assistant. The field calibration values (from both the deployment and retrieval) should be used here. Additional multiparameter measurements can be taken when on site for other surveys throughout the period of deployment.

10.0 Waste disposal

Dispose of sodium sulfite solution by diluting with fresh water and pouring down a sink drain. Refer to the attached MSDS for more information regarding the solution (Attachment B.5).

11.0 References

Onset. 2014. HOBO® Dissolved Oxygen Logger (U26-001) Manual. Bourne, MA: Onset Computer Corporation

<http://www.onsetcomp.com/files/manual_pdfs/15603-B-MAN-U26x.pdf> Accessed 12 Jul 2015.

Onset. 2015. HOBOware® User's Guide. Bourne, MA: Onset Computer Corporation <http://www.onsetcomp.com/files/manual_pdfs/12730-S%20HOBOware%20User%27s%20Guide.pdf> Accessed 19 May 2015.

Attachment B.1

Dissolved Oxygen Data Logger Data Calibration Data Sheet

DISSOLVED OXYGEN DATA LOGGER CALIBRATION DATA SHEET

Sonde Manufacturer:
Model:
Logger ID:

Sensor Cap:

Date Initialized (Installed)	Date Expired

Calibration:

Date	Not from Logger	From Logger Calibration				
	Pressure (mmHg)	100% - Logger	100% - Calculated	0% - Logger	Gain (100%)	Offset (0%)

General logger condition notes:

For troubleshooting refer to logger's user manual.

Attachment B.2

Multiparameter Sonde Calibration Data Sheet

MULTIPARAMETER SONDE CALIBRATION DATA SHEET

Sonde Manufacturer:	
Model:	
Sonde ID:	
Date:	**Time:**
Technician:	

Basic Maintenance and Care:

Check sonde date/time (OK?): YES / NO	
Battery Volts: _______V (change if probe displays battery error message)	Changed batteries: YES / NO
DO membrane changed? YES / NO	

Non-calibrated values:

Parameter	Value
Temperature (ºC)	
Pressure (mmHg)	

Values from calibration:

Parameter	Pre-Calibration	Calibrated Value
D.O. concentration (mg/L)		
D.O. percent saturation (%)		
pH 4		
pH 7		
pH 10		
Conductivity (µs/cm)		

General sonde condition notes:

For troubleshooting refer to sonde's user manual.

Attachment B.3

Continuous Dissolved Oxygen Deployment and Retrieval Data Sheet

CONTINUOUS DISSOLVED OXYGEN DEPLOYMENT AND RETRIEVAL DATA SHEET

(FRONT)

Waterbody Name:	**Date**:
Crew:	
Address:	
Driving Directions/Parking:	
Logging Start Date / Time:	**Logging Interval**:

<table>
<tr><th>Sample Location Code</th><th>Lat / Long</th><th>Site Description</th><th>Stream Parameters</th><th>Multiparameter Readings</th><th>Densio -meter</th><th>Photos</th></tr>
<tr><td></td><td></td><td></td><td>Logger ID #
Wetted width (m) =
Depth (cm) =
MRPD (cm) =
Time in =

Circle as applicable:

Lake / Stream
Riffle / Pool / Run

Streambed geology:</td><td><table><tr><th></th><th>Deploy</th><th>Retrieve</th></tr><tr><td>Date</td><td></td><td></td></tr><tr><td>Time</td><td></td><td></td></tr><tr><td>DO conc</td><td></td><td></td></tr><tr><td>DO %</td><td></td><td></td></tr><tr><td>Temp</td><td></td><td></td></tr><tr><td>pH</td><td></td><td></td></tr><tr><td>Cond</td><td></td><td></td></tr></table></td><td>US:
RR:
DS:
RL:</td><td></td></tr>
<tr><td></td><td></td><td></td><td>Logger ID #
Wetted width (m) =
Depth (cm) =
MRPD (cm) =
Time in =

Circle as applicable:

Lake / Stream
Riffle / Pool / Run

Streambed geology:</td><td><table><tr><th></th><th>Deploy</th><th>Retrieve</th></tr><tr><td>Date</td><td></td><td></td></tr><tr><td>Time</td><td></td><td></td></tr><tr><td>DO conc</td><td></td><td></td></tr><tr><td>DO %</td><td></td><td></td></tr><tr><td>Temp</td><td></td><td></td></tr><tr><td>pH</td><td></td><td></td></tr><tr><td>Cond</td><td></td><td></td></tr></table></td><td>US:
RR:
DS:
RL:</td><td></td></tr>
<tr><td></td><td></td><td></td><td>Logger ID #
Wetted width (m) =
Depth (cm) =
MRPD (cm) =
Time in =

Circle as applicable:

Lake / Stream
Riffle / Pool / Run

Streambed geology:</td><td><table><tr><th></th><th>Deploy</th><th>Retrieve</th></tr><tr><td>Date</td><td></td><td></td></tr><tr><td>Time</td><td></td><td></td></tr><tr><td>DO conc</td><td></td><td></td></tr><tr><td>DO %</td><td></td><td></td></tr><tr><td>Temp</td><td></td><td></td></tr><tr><td>pH</td><td></td><td></td></tr><tr><td>Cond</td><td></td><td></td></tr></table></td><td>US:
RR:
DS:
RL:</td><td></td></tr>
</table>

CONTINUOUS TEMPERATURE DEPLOYMENT AND RETRIEVAL DATA SHEET

(Back)

Site Sketch:

Attachment B.4

Lake Multiparameter Vertical Profile Data Sheet

LAKE MULTIPARAMETER VERTICAL PROFILE DATA SHEET

(Front)

Waterbody Name:	**Date**:	**Crew**:
Lake Depth:	**Sampling Interval (circle one)** 0.5m / 1.0m	

GPS Lat:	**GPS Long**:
Site description:	

Depth (m)	**Recorded Parameters**			
	Temperature (°C)	**DO concentration (mg/L)**	**DO % Saturation**	**Conductivity (µs/cm)**
Surface (0.1 m)				

Notes:

LAKE MULTIPARAMETER VERTICAL PROFILE DATA SHEET

(Back)

Site Sketch:

Attachment B.5

Material Safety Data Sheet (MSDS) for Onset© Sodium Sulfite Solution

onset

HOBO Data Loggers

Material Safety and Data Sheet

I. Chemical Product and Company Identification

Product Name:	**Manufacturer:**	**Emergency Contact:**
Sodium Sulfite **2 Molar Standard Solution**	Aqua Phoenix Scientific, Inc. 320 Maple Ave. Hanover, PA 17331 Telephone: 866-632-1291 Fax: 717-633-1285	INFOTRAC Emergency Response Hotline: 1-800-535-5053 (in the U.S. and Canada) 1-352-323-3500 www.infotrac.net

II. Composition, Information on Ingredients

Hazardous Components Specific Chemical Identity: Common Names	CAS NO.	%	OSHA PEL	ACGIH TLV
Sodium Sulfite	7757-83-7	5% w/v	Not listed	Not Established
Cobalt Chloride Hexahydrate	7791-13-1	<0.01% w/v	Not listed	0.1 mg/m^3

III. Hazard Identification

Emergency Overview: Non-flammable, non-corrosive. Does not present significant health hazards. Wash areas of contact with water.

Target Organs: Eyes, Skin

Eyes	May cause slight irritation
Skin	May cause slight irritation
Ingestion	Large doses may cause upset stomach
Inhalation	Not likely to be a hazard
Chronic Effect /Carcinogenicity	None (IARC, NTP, OSHA)

IV. First Aid

Eyes	Immediately flush eyes with water for at least 15 minutes. Immediately get medical assistance.
Skin	Flush with water for 15 minutes. Get medical assistance if irritation develops.
Ingestion	DO NOT induce vomiting. Dilute with water or milk. Get medical assistance.
Inhalation	Remove to fresh air. Give artificial respiration if necessary. If breathing is difficult, give oxygen.

V. Fire Fighting Measures

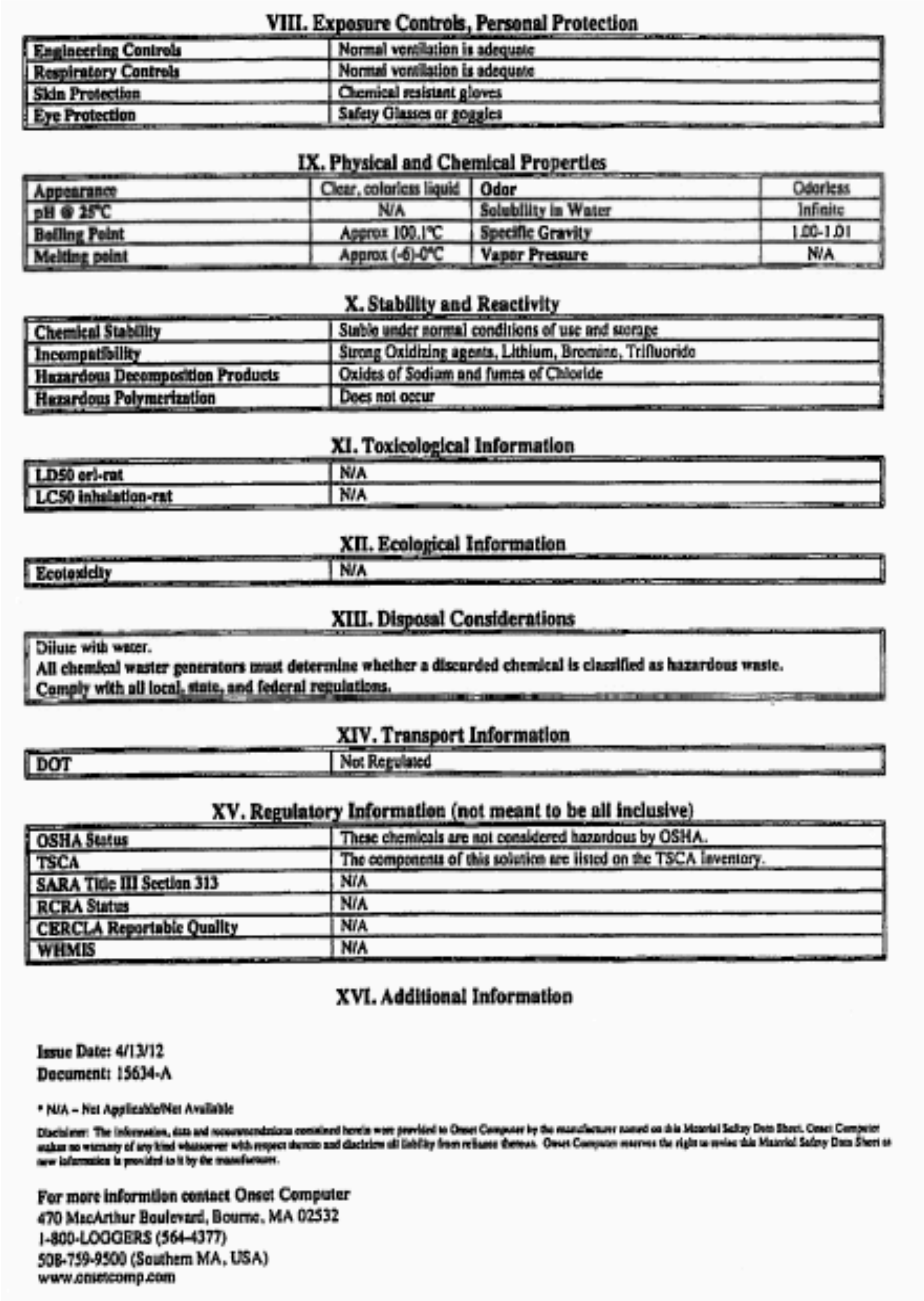
Flash Point	N/A
Extinguishing Media	Use means suitable to extinguishing surrounding fire.
Fire and Explosion Hazards	Not considered to be a fire or explosion hazard.
Fire Fighting Instructions/Equipment	Use normal procedures. Poisonous gases may be produced in fire. Use protective clothing. Use NIOSH-approved breathing equipment.
NFPA Rating	(estimated) Health: 1, Flammable: 0, Reactivity: 0

VI. Accidental Release Measures

Absorb with suitable material. Always obey local regulations.

VII. Handling and Storage

Handling	Wash hands after handling. Avoid contact with skin and eyes.
Storage	Protect from freezing and physical damage.



VIII. Exposure Controls, Personal Protection

Engineering Controls	Normal ventilation is adequate
Respiratory Controls	Normal ventilation is adequate
Skin Protection	Chemical resistant gloves
Eye Protection	Safety Glasses or goggles

IX. Physical and Chemical Properties

Appearance	Clear, colorless liquid	**Odor**	Odorless
pH @ 25°C	N/A	**Solubility in Water**	Infinite
Boiling Point	Approx 100.1°C	**Specific Gravity**	1.00-1.01
Melting point	Approx (-6)-0°C	**Vapor Pressure**	N/A

X. Stability and Reactivity

Chemical Stability	Stable under normal conditions of use and storage
Incompatibility	Strong Oxidizing agents, Lithium, Bromine, Trifluoride
Hazardous Decomposition Products	Oxides of Sodium and fumes of Chloride
Hazardous Polymerization	Does not occur

XI. Toxicological Information

LD50 orl-rat	N/A
LC50 inhalation-rat	N/A

XII. Ecological Information

Ecotoxicity	N/A

XIII. Disposal Considerations

Dilute with water.
All chemical waster generators must determine whether a discarded chemical is classified as hazardous waste.
Comply with all local, state, and federal regulations.

XIV. Transport Information

DOT	Not Regulated

XV. Regulatory Information (not meant to be all inclusive)

OSHA Status	These chemicals are not considered hazardous by OSHA.
TSCA	The components of this solution are listed on the TSCA Inventory.
SARA Title III Section 313	N/A
RCRA Status	N/A
CERCLA Reportable Quality	N/A
WHMIS	N/A

XVI. Additional Information

Issue Date: 4/13/12
Document: 15634-A

* N/A – Not Applicable/Not Available

Disclaimer: The information, data and recommendations contained herein were provided to Onset Computer by the manufacturer named on this Material Safety Data Sheet. Onset Computer makes no warranty of any kind whatsoever with respect thereto and disclaims all liability from reliance thereon. Onset Computer reserves the right to revise this Material Safety Data Sheet as new information is provided to it by the manufacturer.

For more informtion contact Onset Computer
470 MacArthur Boulevard, Bourne, MA 02532
1-800-LOGGERS (564-4377)
508-759-9500 (Southern MA, USA)
www.onsetcomp.com

APPENDIX C

MONITORING LOCATIONS

Latitude and longitudes for each of the temperature logger deployment locations in this study.

Site	Logger Name	Latitude	Longitude
AMU	AMUUS	42.381835	-72.438160
	AMUDS3	42.380001	-72.451060
AHA	AHADS5	42.379904	-72.451076
AML	AMLDS3	42.381016	-72.459128
	AMLDS4	42.381099	-72.459731
BAL	BALUS	42.664950	-71.145800
	BALIMP	42.671867	-71.149417
	BALDS1	42.672655	-71.149479
	BALDS2	42.673552	-71.149397
	BALDS3	42.676033	-71.149667
	BALDS4	42.677458	-71.149530
BAR	BARUS	41.876633	-71.052533
	BARIMP	41.882232	-71.048601
	BARDS1	41.882483	-71.048100
	BARDS2	41.882779	-71.047658
	BARDS3	41.882883	-71.046967
	BARDS4	41.883091	-71.045786
	BARDS5	41.883583	-71.044683
BOS	BOSUS	42.573517	-71.054133
	BOSIMP	42.570100	-71.031250
	BOSDS1	42.569949	-71.030646
	BOSDS2	42.569875	-71.029418
	BOSDS3	42.569583	-71.026667
	BOSDS4	42.569317	-71.024450
	BOSDS5	42.568083	-71.019367
BVL	BVLUS	42.595117	-71.195383
	BVLIMP	42.626883	-71.158050
	BVLDS1	42.627478	-71.157397
	BVLDS2	42.629133	-71.157383
	BVLDS3	42.631017	-71.158150
CGM	CGMUS	42.022000	-70.950650
	CGMIMP	42.021500	-70.950883
	CGMDS1	42.021254	-70.951298
	CGMDS2	42.020383	-70.952217
	CGMDS3	42.019395	-70.952379
	CGMDS4	42.016883	-70.953817
	CGMDS5	42.013783	-70.954433

Site	Logger Name	Latitude	Longitude
CRA	CRAUS	42.494789	-72.523456
	CRADS1	42.503325	-72.524879
	CRADS2	42.503987	-72.525017
	CRADS3	42.504920	-72.525590
	CRADS4	42.506620	-72.526899
	CRADS5	42.504917	-72.525621
EBB	EBBDS1	42.483659	-71.933326
	EBBDS2	42.482876	-71.936248
	EBBDS3	42.478570	-71.941508
HUN	HUNUS	42.219704	-70.787035
	HUNIMP	42.222867	-70.788567
	HUNDS1	42.223250	-70.788733
	HUNDS2	42.223467	-70.788750
IPS	IPSUS	42.658178	-70.861826
	IPSIMP	42.676767	-70.838150
	IPSDS1	42.678200	-70.837583
	IPSDS2	42.679524	-70.837307
LAR	LARDS1	42.320375	-73.329566
	LARDS2	42.320178	-73.330300
	LARDS3	42.318400	73.330470
	LARDS4	42.316040	-73.332119
	LARDS5	42.312056	-73.332903
LRM	LRMDS1	42.353850	-72.707233
	LRMDS2	42.353867	-72.706050
	LRMDS3	42.354050	-72.703117
	LRMDS4	42.354200	-72.702183
MAR	MARUS	42.656892	-71.146506
	MARIMP	42.662050	-71.146683
	MARDS1	42.662717	-71.146733
	MARDS2	42.663229	-71.146659
	MARDS3	42.664183	-71.146500
MOO	MOODS1	42.190560	-72.811050
	MOODS2	42.187380	-72.809940
	MOODS3	42.185953	-72.810237
	MOODS4	42.185422	-72.810587
MRM	MRMUS2	42.352767	-72.714417
	MRMIMP	42.351433	-72.710800
	MRMDS	42.352200	-72.709500
MUN	MUNDS1	42.083493	-72.842247
	MUNDS2	42.082247	-72.841898
	MUNDS3	42.081690	-72.840380

Site	Logger Name	Latitude	Longitude
OLD	OLDUS	42.122600	-71.447867
	OLDIMP	42.130708	-71.444343
	OLDDS1	42.131128	-71.443899
	OLDDS2	42.131733	-71.443300
	OLDDS3	42.133233	-71.442433
	OLDDS4	42.134583	-71.442000
	OLDDS5	42.135050	-71.441183
PEC	PECIMP	42.476460	-73.267953
	PECDS1	42.476314	-73.266712
	PECDS2	42.476000	-73.266177
	PECDS3	42.473323	-73.263862
	PECDS4	42.471967	-73.259400
PIC	PICDS1	42.244420	-71.602926
	PICDS2	42.244777	-71.600469
	PICDS3	42.245134	-71.599076
	PICDS4	42.252808	-71.591077
PRD	PRDDS1	42.437544	-72.364817
	PRDDS2	42.438078	-72.364941
	PRDDS3	42.438863	-72.365379
	PRDDS4	42.441554	-72.368350
	PRDDS5	42.442148	-72.368790
ROA	ROAUS	42.468821	-72.657030
	ROADS1	42.466576	-72.652658
	ROADS2	42.466365	-72.651964
	ROADS3	42.464456	-72.650177
	ROADS4	42.461702	-72.642676
TEL	TELUS	42.462250	-73.254450
	TELIMP	42.447010	-73.263855
	TELDS1	42.446444	-73.263197
	TELDS2	42.446350	-73.261983
	TELDS3	42.443333	-73.2609
	TELDS4	42.438250	-73.261800
	TELDS5	42.437450	-73.260567
TUR	TURUS	42.690852	-71.589829
	TURIMP	42.676024	-71.582277
	TURDS1	42.674600	-71.581383
	TURDS2	42.674433	-71.581050
	TURDS3	42.672517	-71.579167
	TURDS4	42.671783	-71.576400
	TURDS5	42.673223	-71.564796

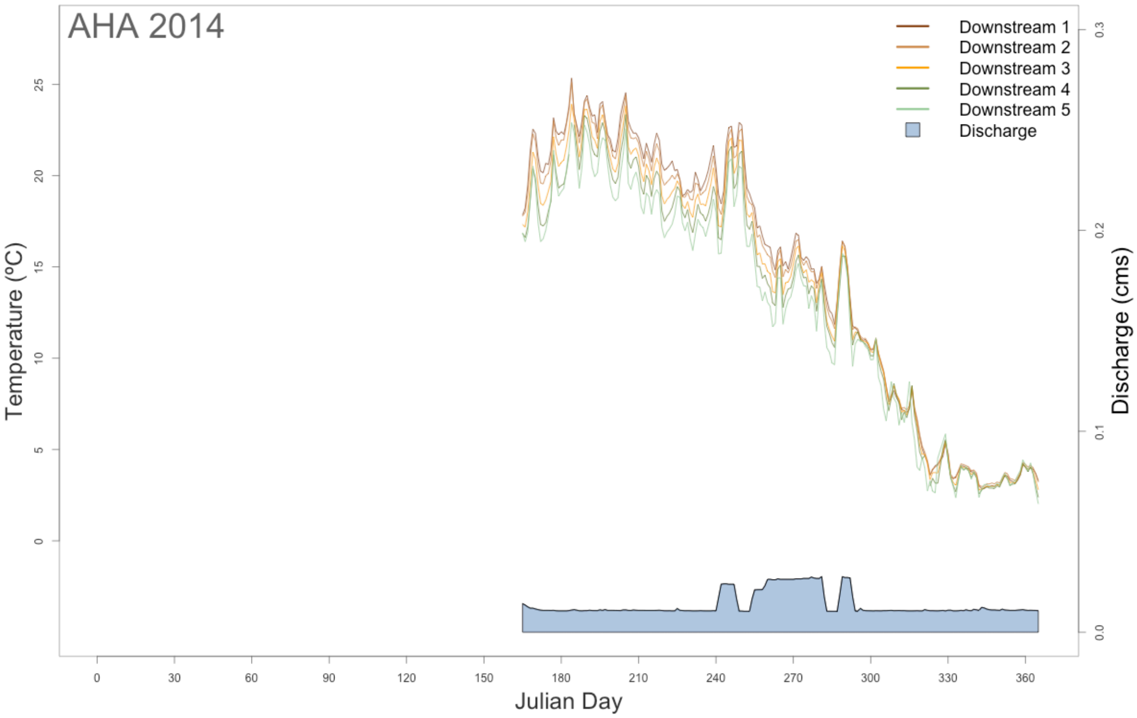
Site	Logger Name	Latitude	Longitude
UND	UNDDS1	42.452373	-72.341434
	UNDDS2	42.449266	-72.337749
	UNDDS3	42.4484975	-72.336057
	UNDDS4	42.4455921	-72.332799
	UNDDS5	42.4454258	-72.332134
UNT	UNTDS1	42.4427343	-72.336988
	UNTDS2	42.4432173	-72.335315
	UNTDS3	42.4446562	-72.334089
UPN	UPNDS1	42.659519	-71.937554
	UPNDS3	42.660042	-71.938420
	UPNDS4	42.660450	-71.939136
URM	URMUS	42.337417	-72.734517
	URMIMP	42.338132	-72.728243
	URMDS1	42.338150	-72.727700
	URMDS2	42.338633	-72.726300
	URMDS3	42.339433	-72.724600
	URMDS4	42.340300	-72.720283
	URMDS5	42.348067	-72.713117
WBH	WBHIMP	42.484517	-73.246600
	WBHDS1	42.483940	-73.246200
	WBHDS2	42.483066	-73.246524
	WBHDS3	42.482563	-73.246836
	WBHDS4	42.478759	-73.248440
	WBHDS5	42.477484	-73.246223
	WBHDS6	42.473732	-73.246352

Latitude and longitudes for each of the dissolved oxygen logger deployment locations in this study.

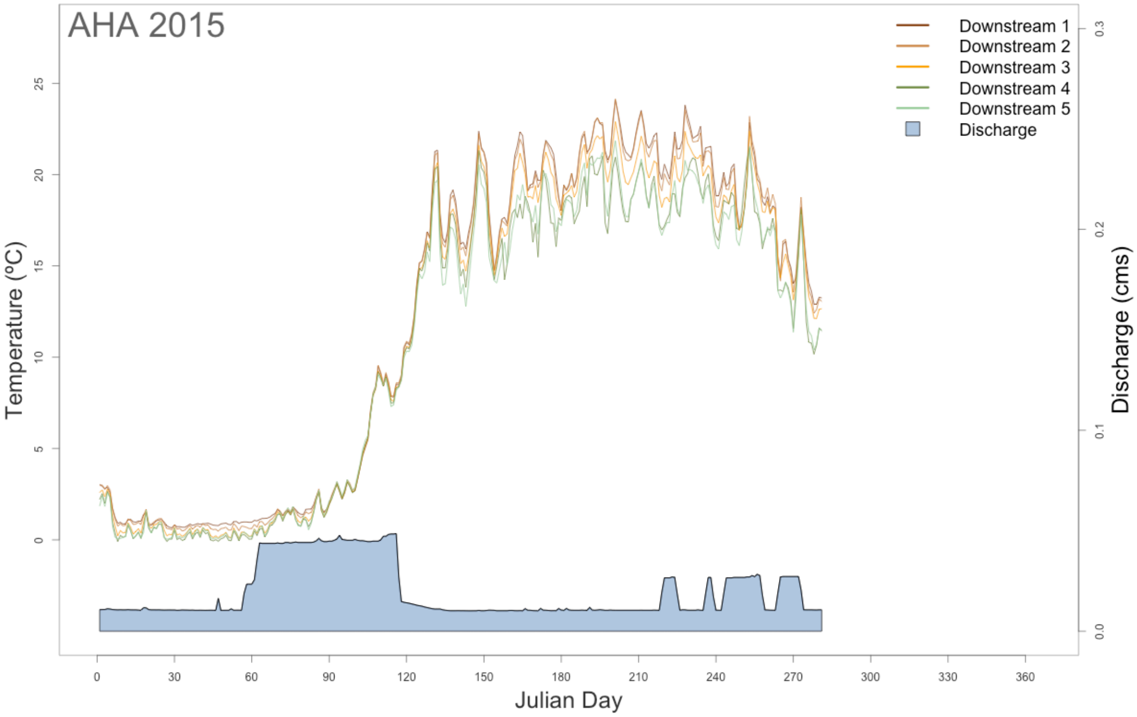
Site	Logger Name	Latitude	Longitude
BAL	BALUS	42.664967	-71.145617
	BALIMP	42.671867	-71.149567
	BALDS	42.672909	-71.149467
BAR	BARUS	41.876609	-71.052462
	BARIMP	41.882232	-71.048601
	BARDS	41.882426	-71.048079
BOS	BOSUS	42.573517	-71.054133
	BOSIMP	42.570100	-71.031250
	BOSDS	42.569931	-71.030376
CGM	CGMUS	42.021992	-70.950556
	CGMIMP	42.021500	-70.951298
	CGMDS	42.021292	-70.951273
HUN	HUNUS	42.219775	-70.787029
	HUNIMP	42.222867	-70.788567
	HUNDS	42.658242	-70.861633
IPS	IPSUS	42.658192	70.861539
	IPSIMP	42.676767	-70.838150
	IPSDS	42.677900	-70.837556
MAR	MARUS	42.656892	-71.146506
	MARIMP	42.662050	-71.146683
	MARDS	42.663133	-71.146735
OLD	OLDUS	42.122600	-71.447867
	OLDIMP	42.130708	-71.444343
	OLDDS	42.131128	-71.443899
RAT	RATUS	41.776471	-71.089094
	RATIMP	41.780650	-71.086450
	RATDS	41.780867	-71.086650
TEL	TELUS	42.462250	-73.254450
	TELIMP	42.447010	-73.263855
	TELDS	42.446441	-73.263119
TUR	TURUS	42.690852	-71.589829
	TURIMP	42.676024	-71.582277
	TURDS	42.674600	-71.581383
URM	URMUS	42.337417	-72.734517
	URMIMP	42.338132	-72.728243
	URMDS	42.338150	-72.727700

APPENDIX D

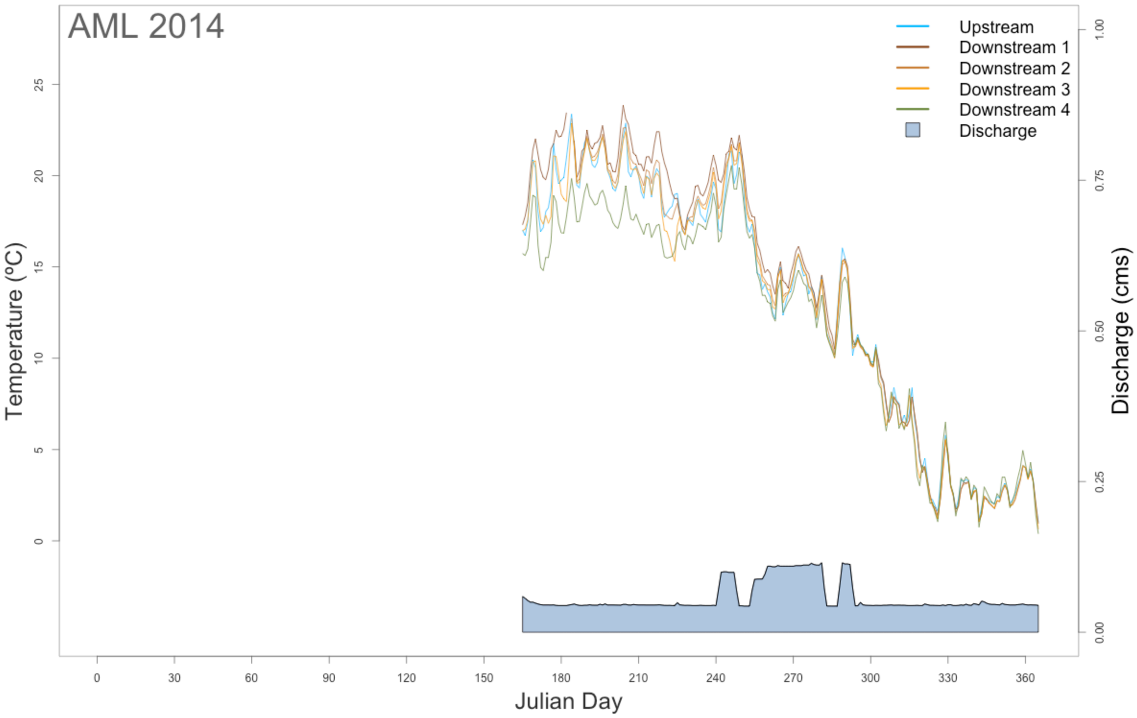
SITE-SPECIFIC MEAN DAILY PRE-DAM REMOVAL TEMPERATURE PROFILES SEPARATED BY YEAR



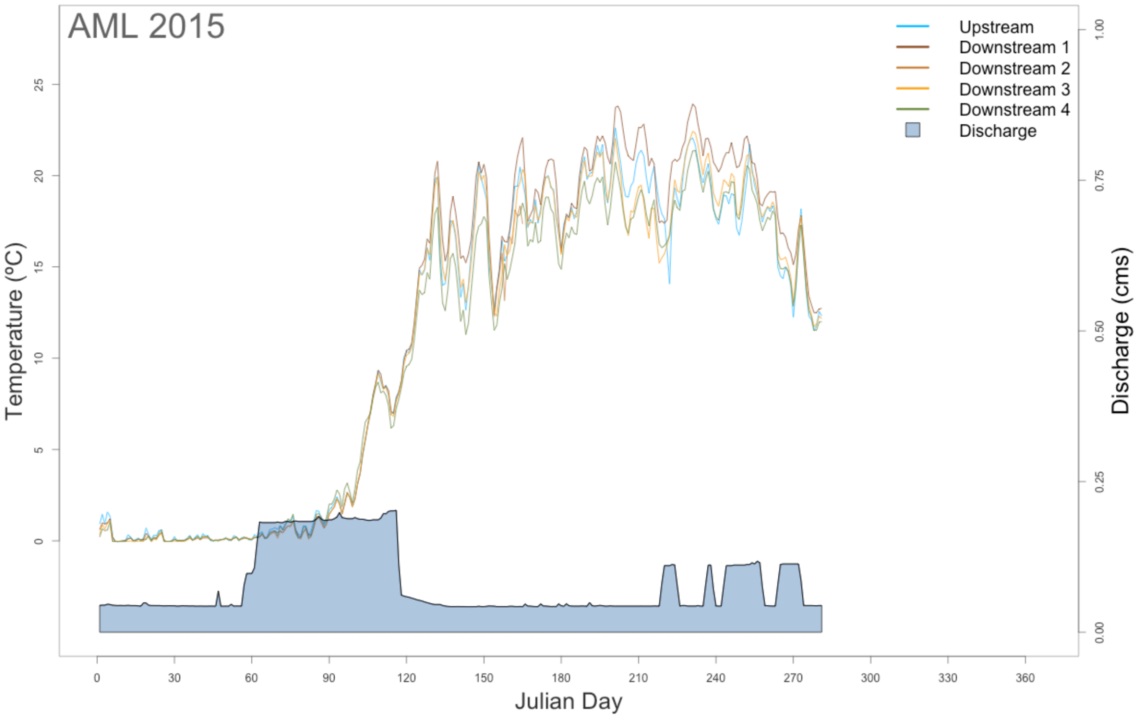
Mean daily temperature from the 5 loggers deployed downstream of Amethyst Brook – Hawley Reservoir Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



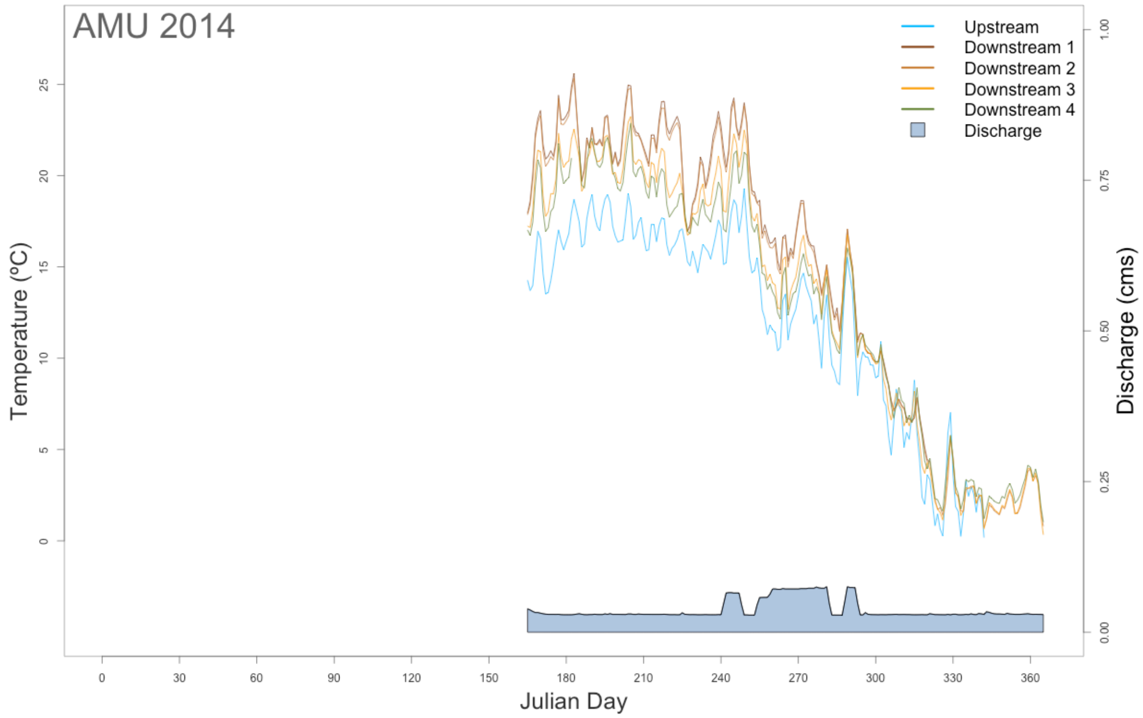
Mean daily temperature from the 5 loggers deployed downstream of Amethyst Brook – Hawley Reservoir Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



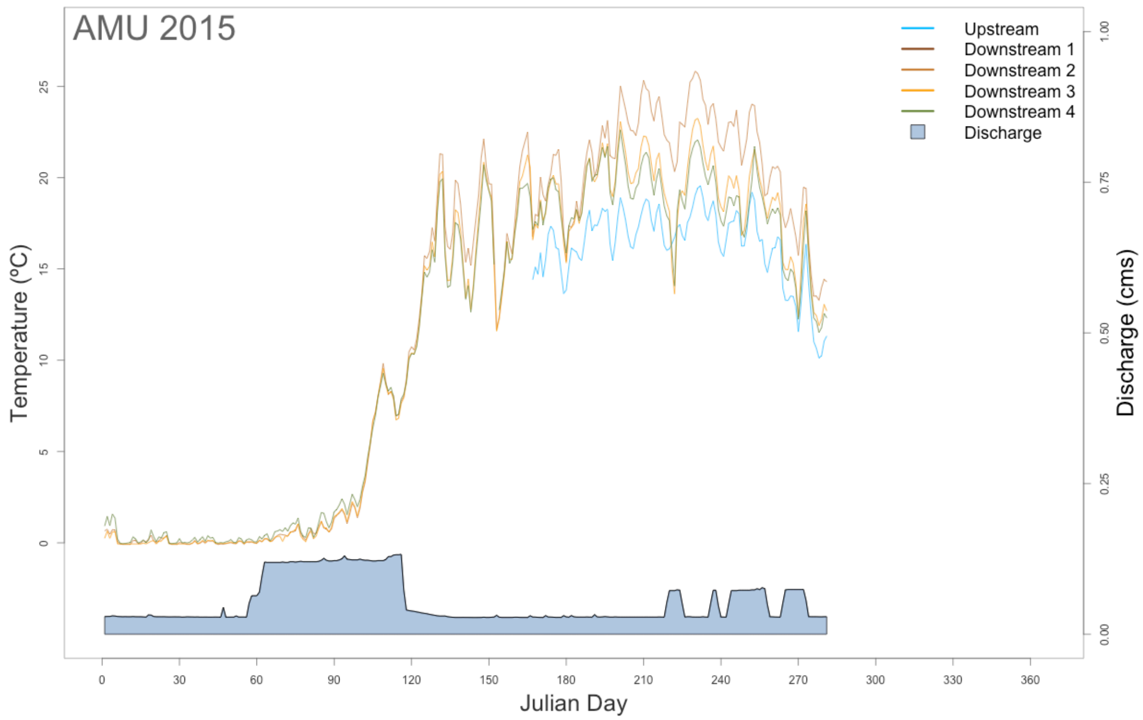
Mean daily temperature from the 5 loggers deployed upstream and downstream of Amethyst Brook – Lower Meetinghouse Reservoir Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



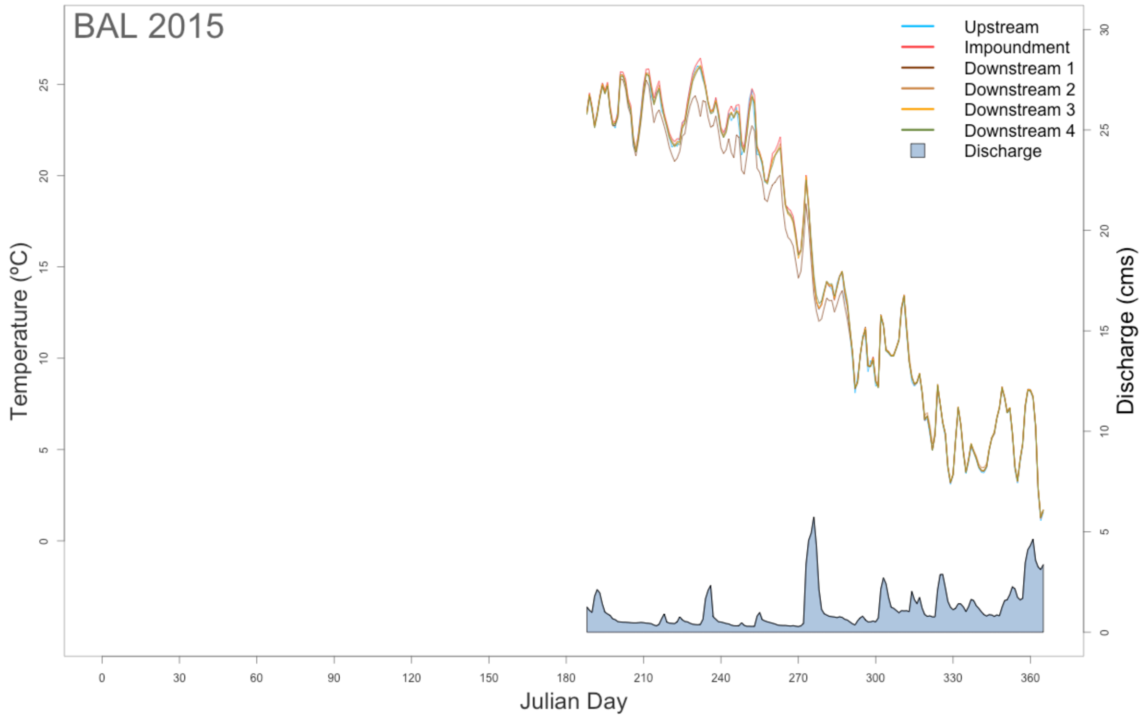
Mean daily temperature from the 5 loggers deployed upstream and downstream of Amethyst Brook – Lower Meetinghouse Reservoir Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



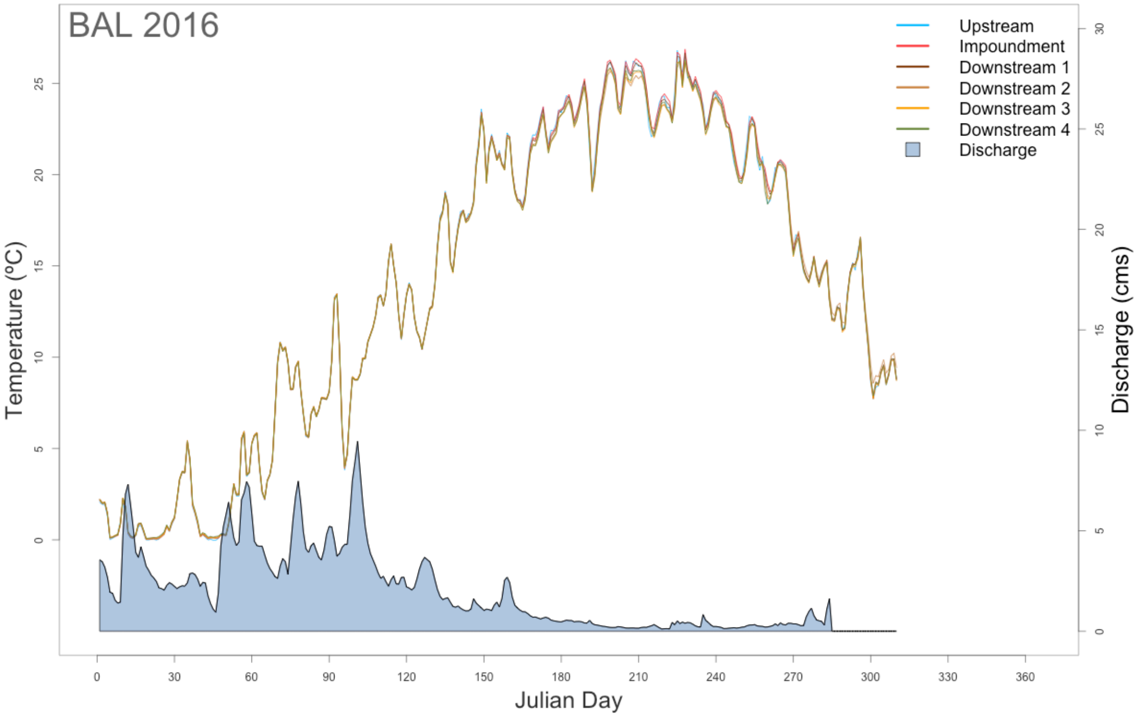
Mean daily temperature from the 5 loggers deployed upstream and downstream of Amethyst Brook – Upper Meetinghouse Reservoir Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



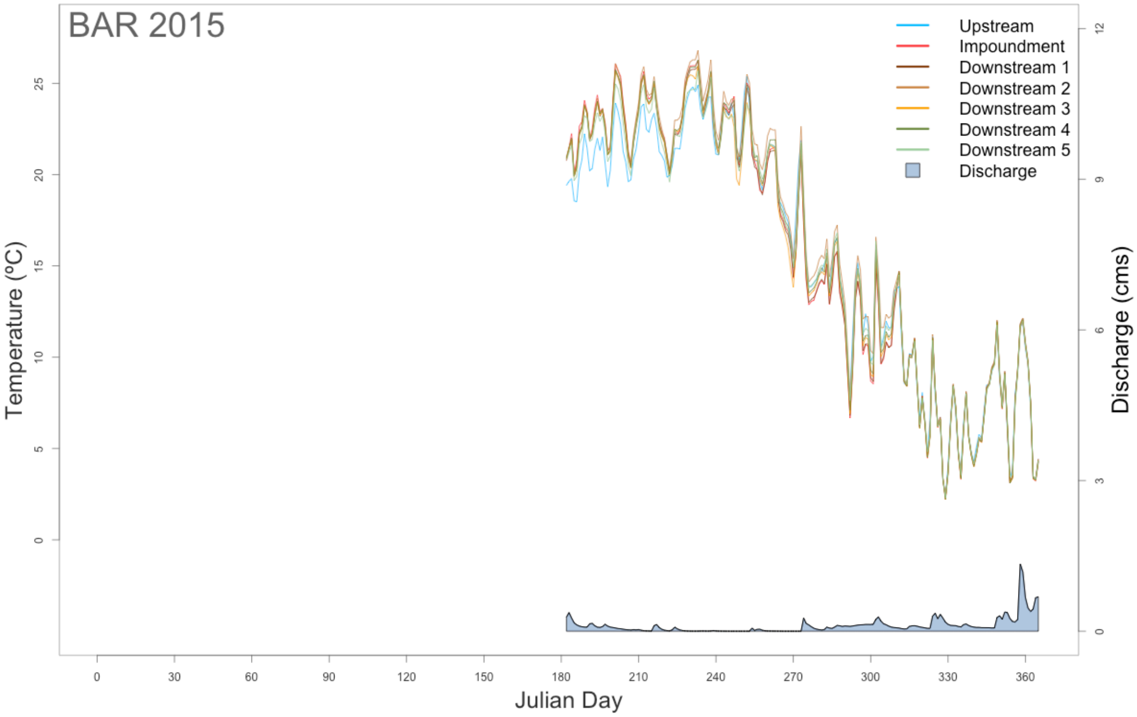
Mean daily temperature from the 5 loggers deployed upstream and downstream of Amethyst Brook – Upper Meetinghouse Reservoir Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



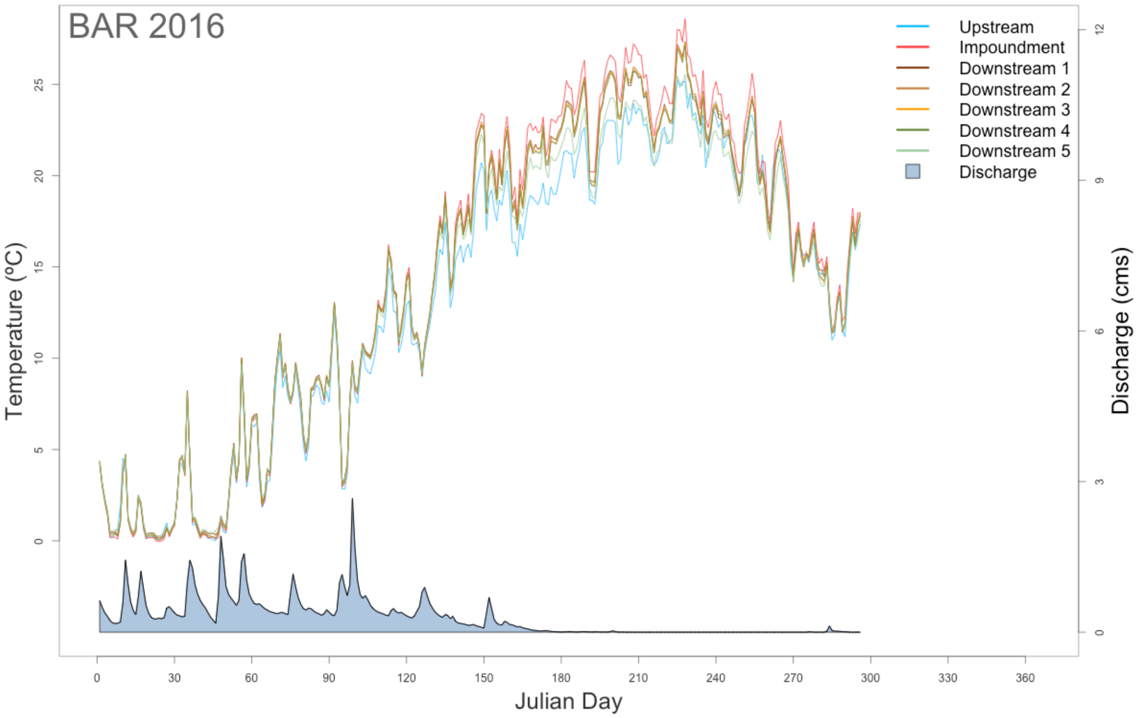
Mean daily temperature from the 6 loggers deployed upstream, within the impoundment, and downstream of Balmoral Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



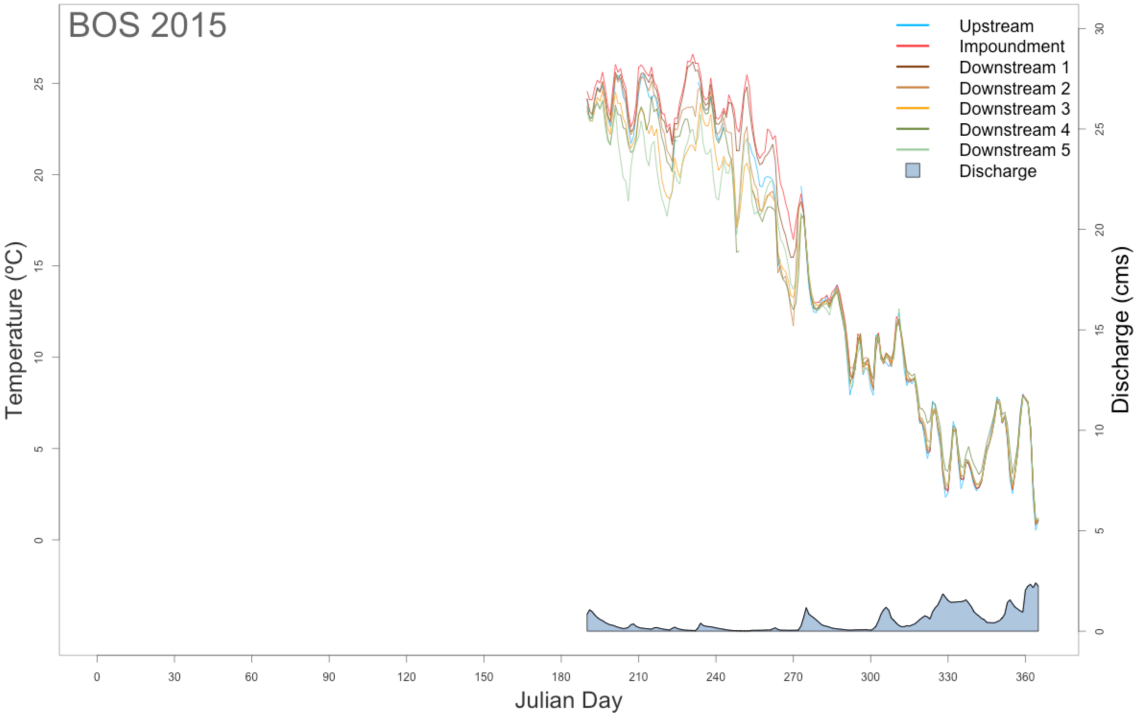
Mean daily temperature from the 6 loggers deployed upstream, within the impoundment, and downstream of Balmoral Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



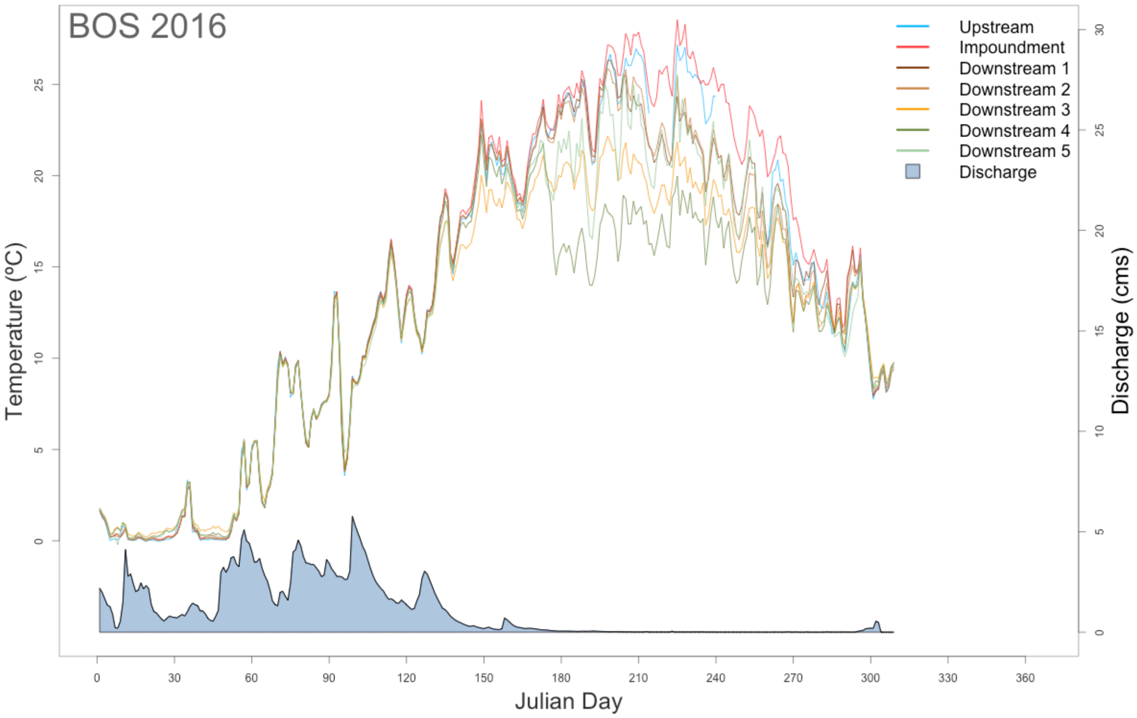
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Barstow's Pond Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



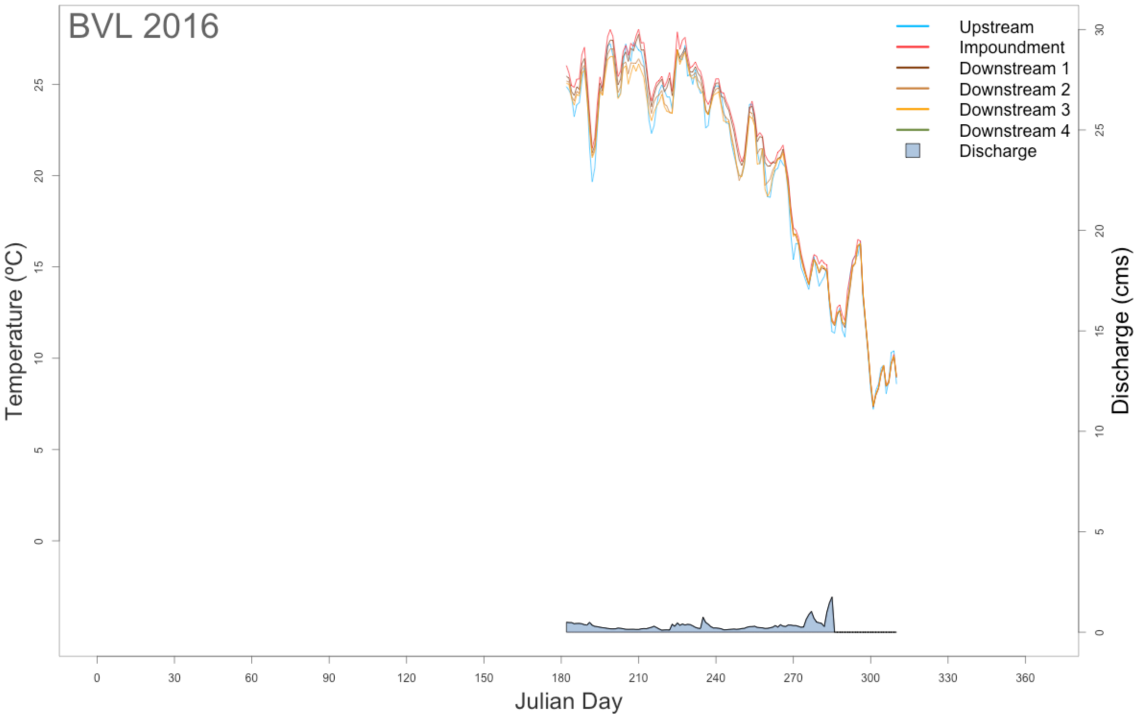
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Barstow's Pond Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



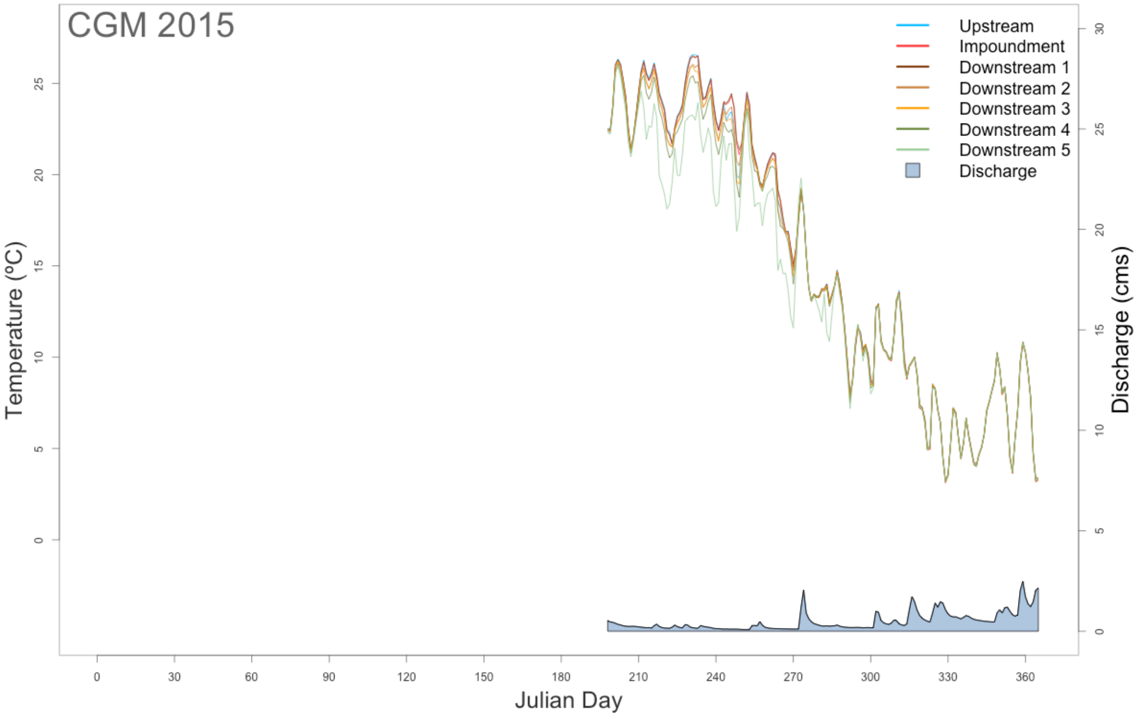
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Bostik / South Middleton Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



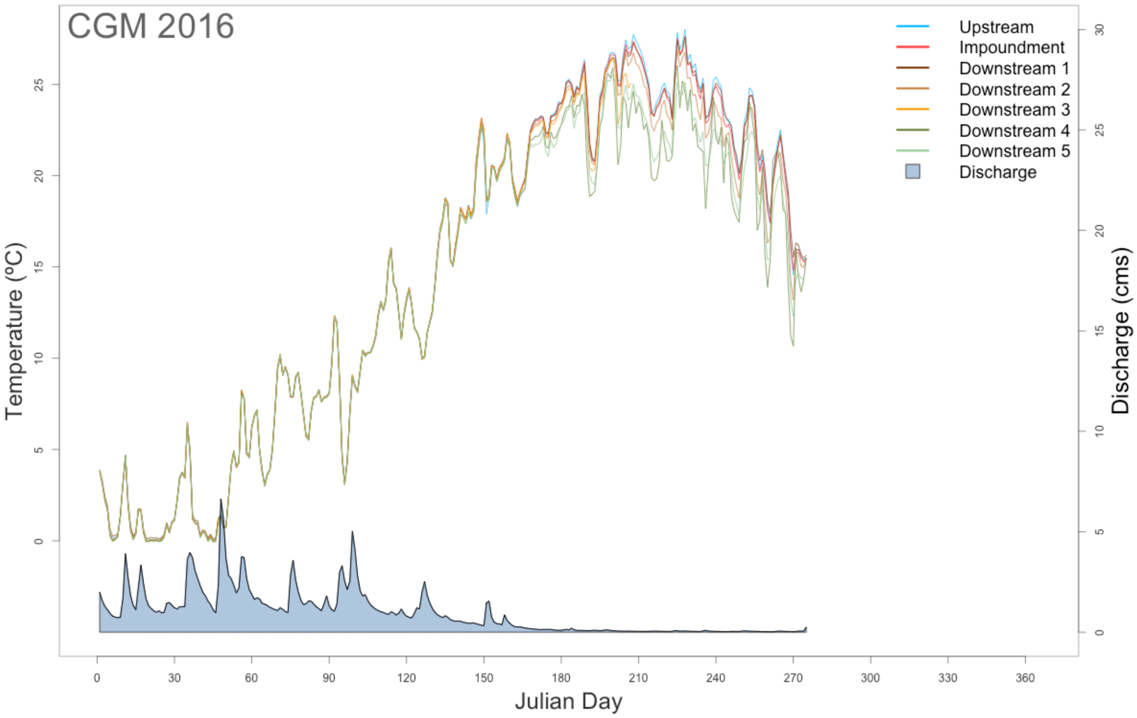
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Bostik / South Middleton Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



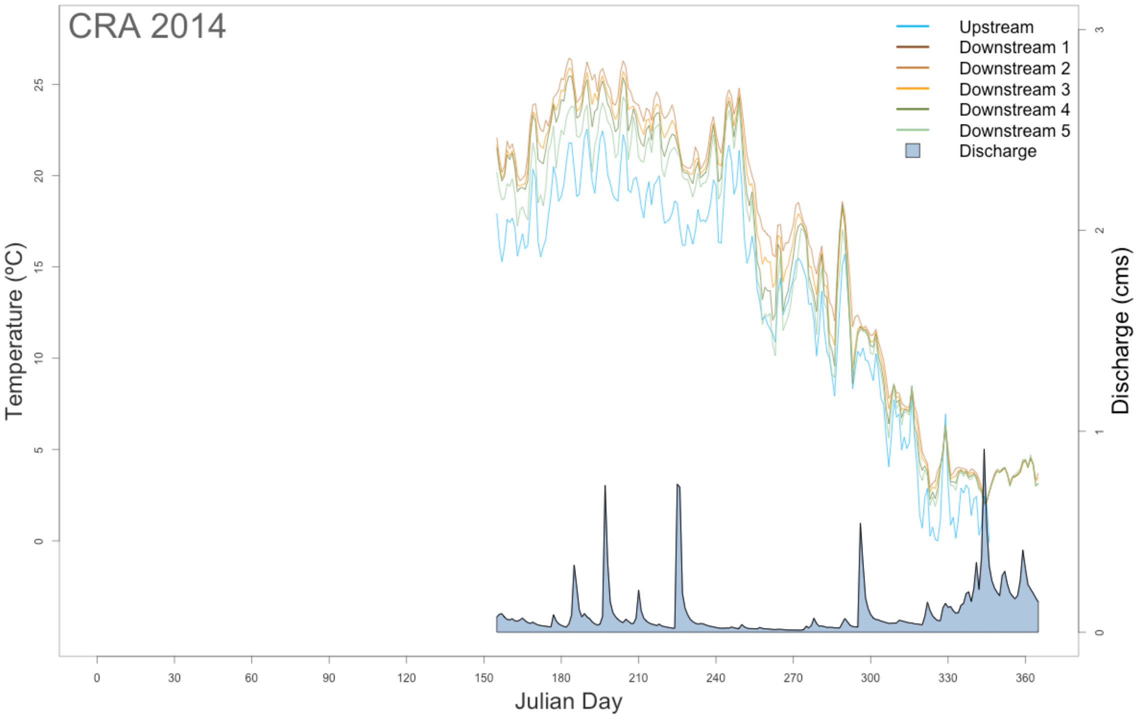
Mean daily temperature from the 5 loggers deployed upstream, within the impoundment, and downstream of Ballardvale Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



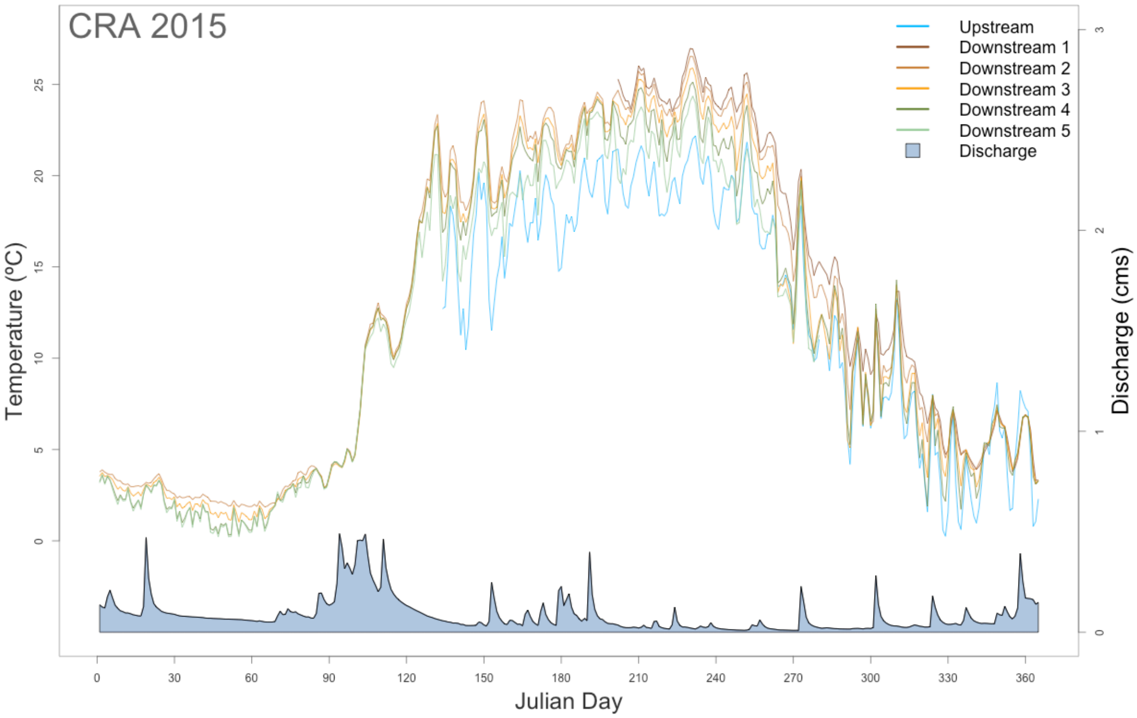
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Cotton Gin Mill Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



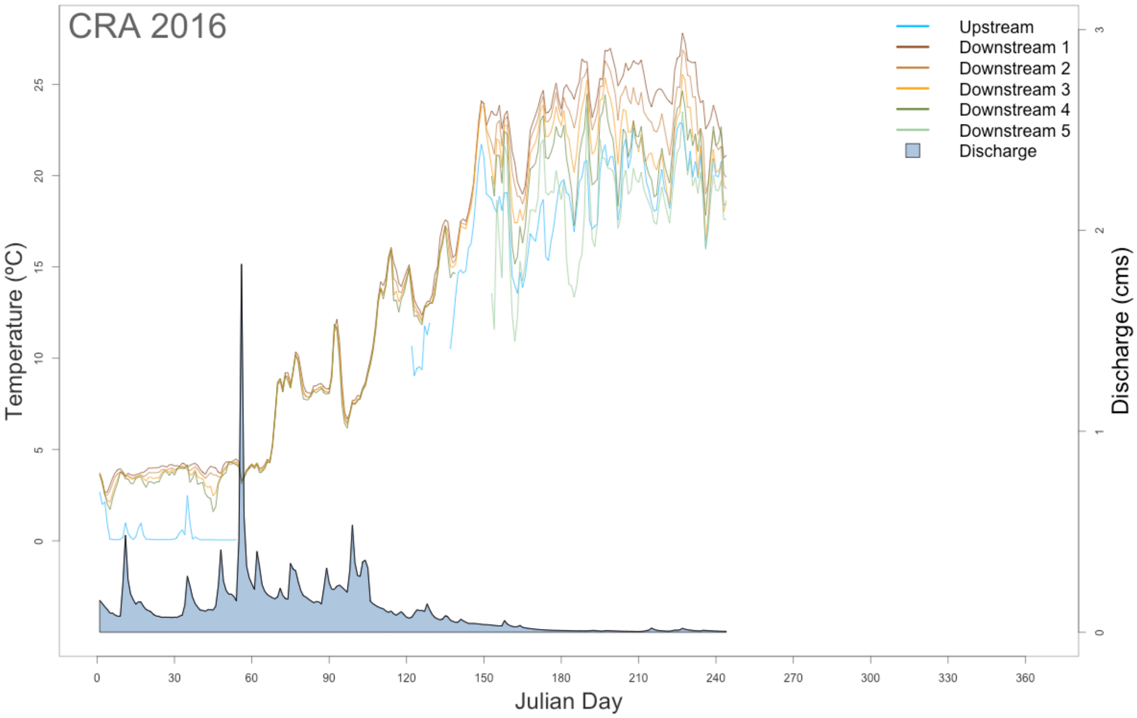
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Cotton Gin Mill Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



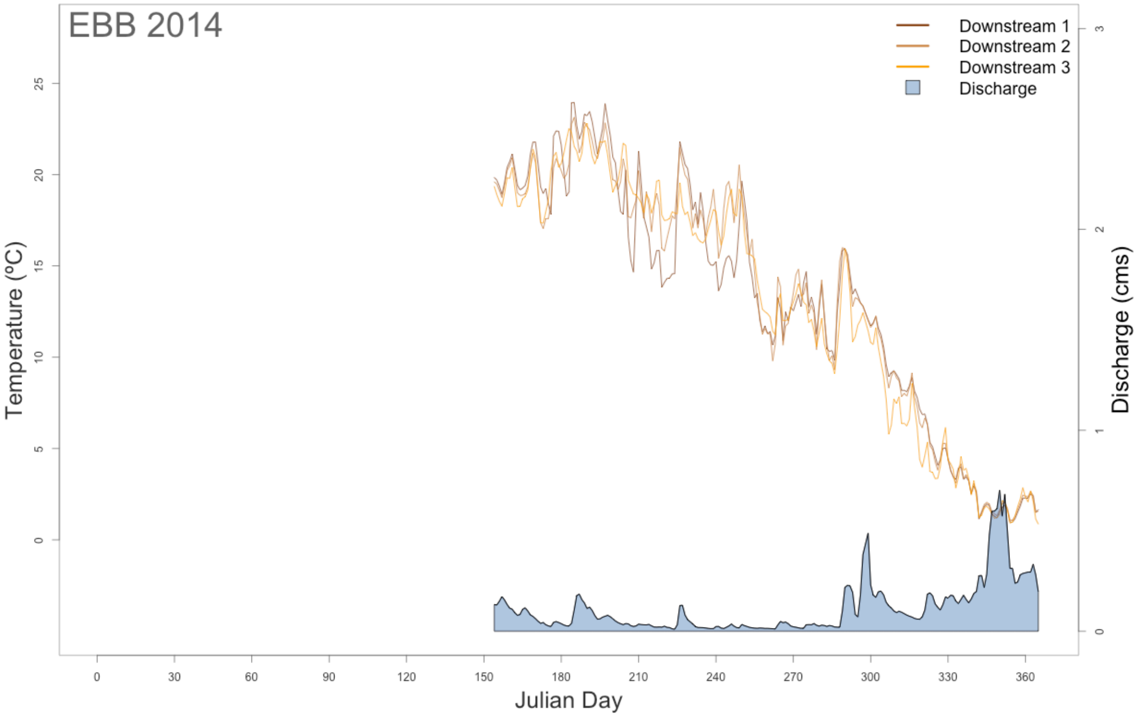
Mean daily temperature from the 6 loggers deployed upstream and downstream of Cranberry Pond Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



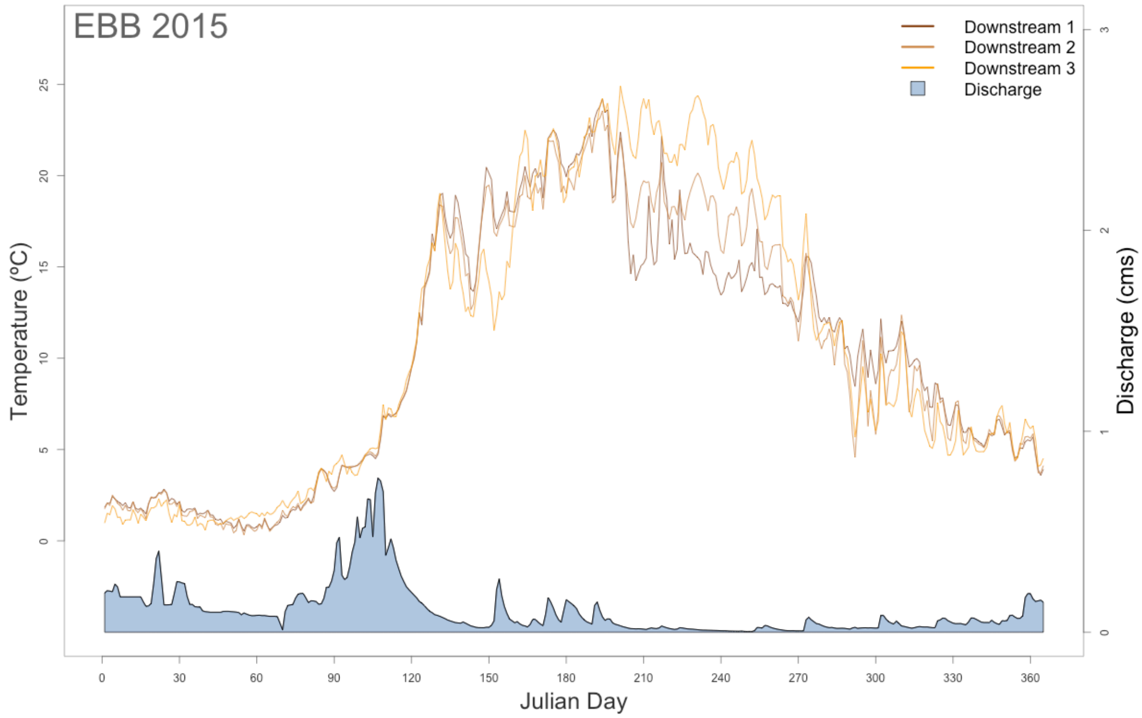
Mean daily temperature from the 6 loggers deployed upstream and downstream of Cranberry Pond Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



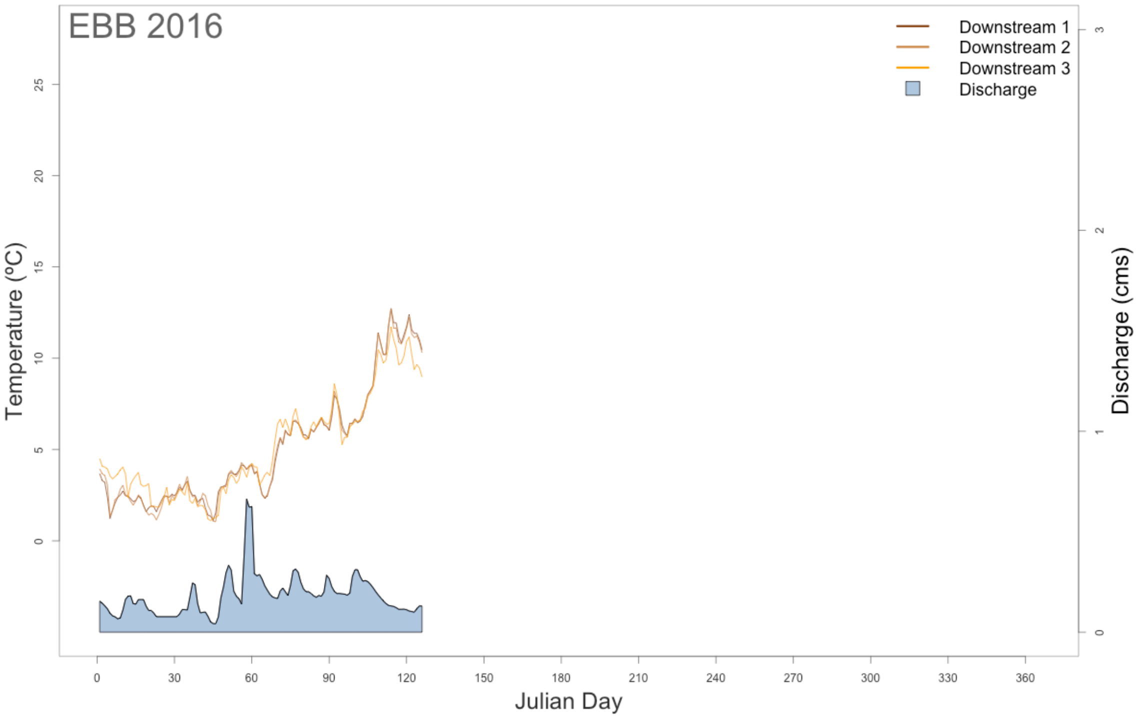
Mean daily temperature from the 6 loggers deployed upstream and downstream of Cranberry Pond Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



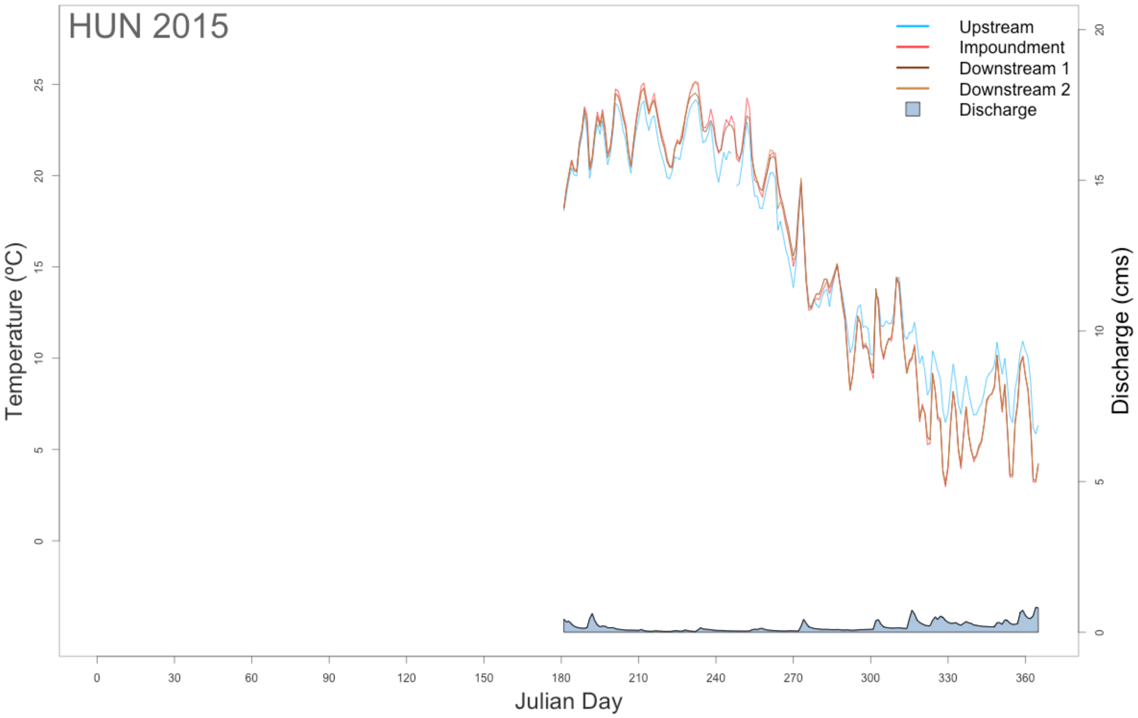
Mean daily temperature from the 3 loggers deployed downstream of East Branch Ware River – Bickford Pond Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



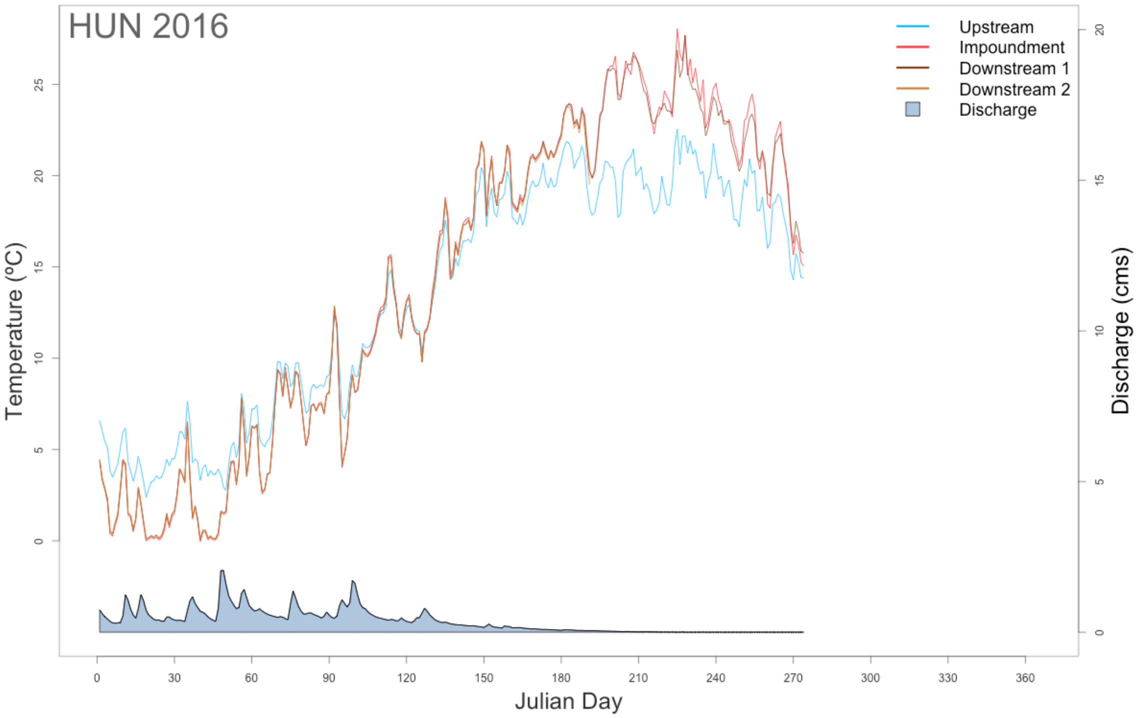
Mean daily temperature from the 3 loggers deployed downstream of East Branch Ware River – Bickford Pond Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



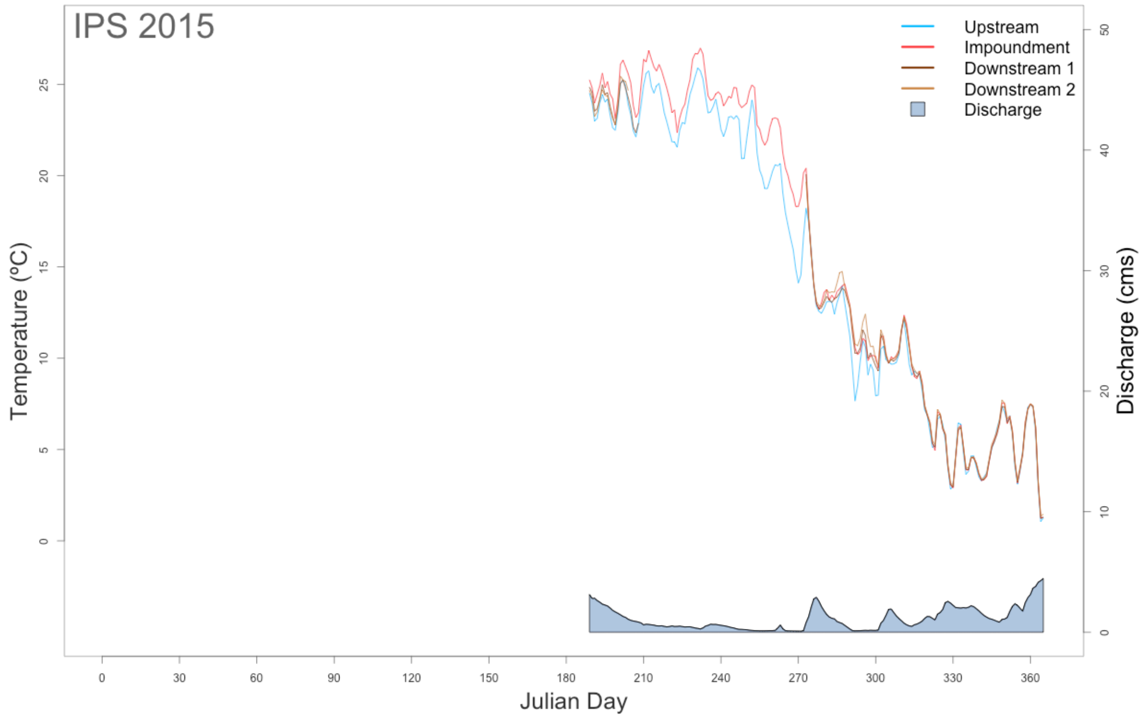
Mean daily temperature from the 3 loggers deployed downstream of East Branch Ware River – Bickford Pond Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



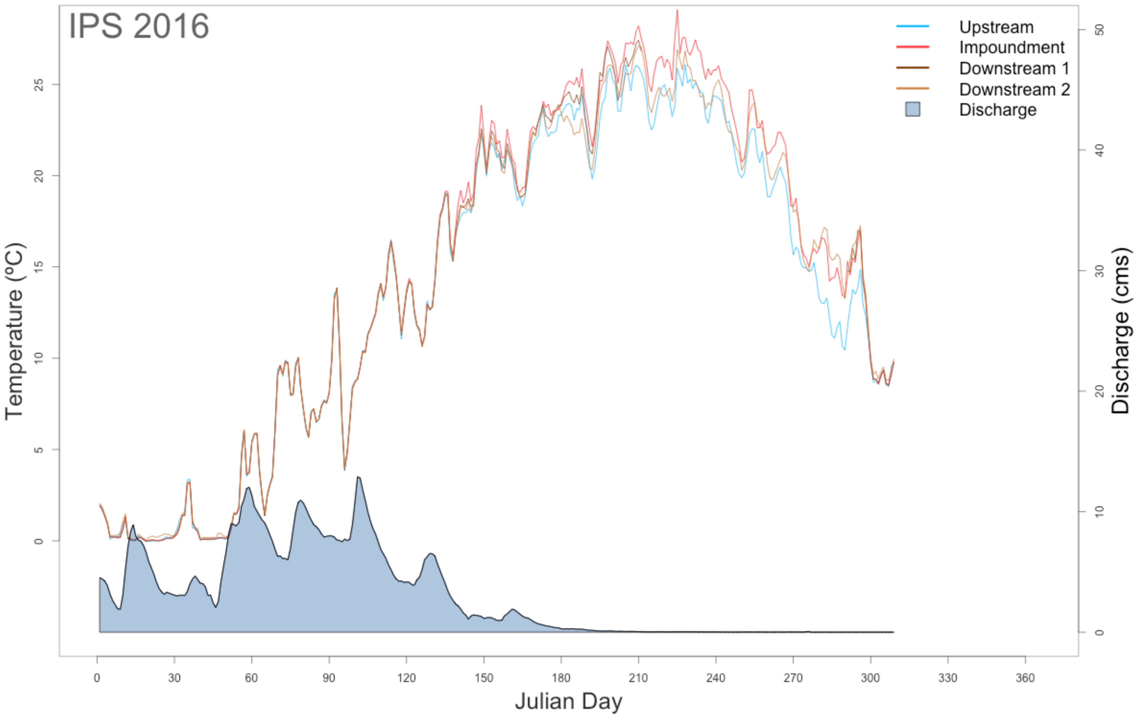
Mean daily temperature from the 4 loggers deployed upstream, within the impoundment, and downstream of Hunter's Pond Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



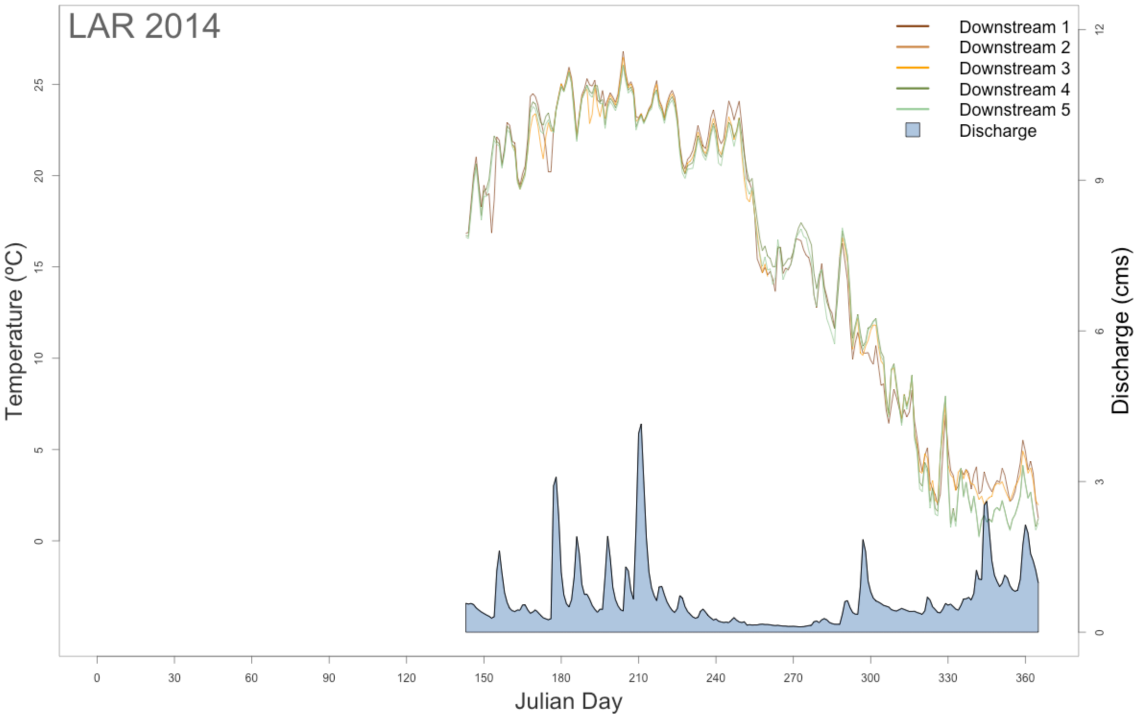
Mean daily temperature from the 4 loggers deployed upstream, within the impoundment, and downstream of Hunter's Pond Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



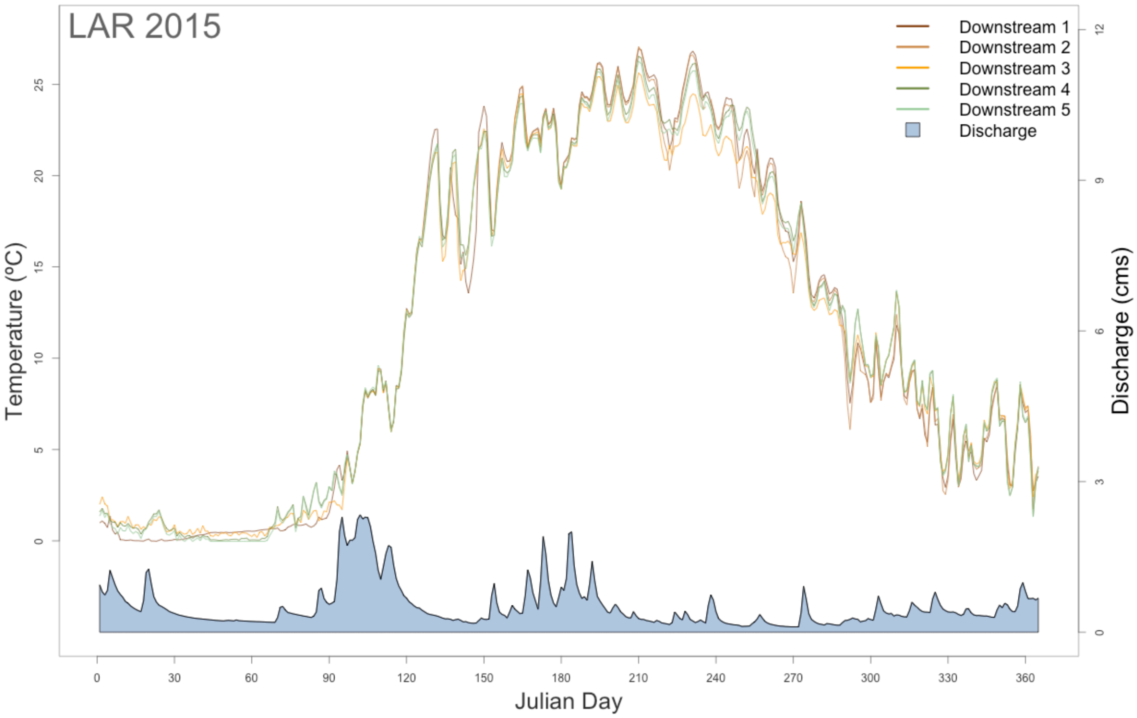
Mean daily temperature from the 4 loggers deployed upstream, within the impoundment, and downstream of Ipswich Mills Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



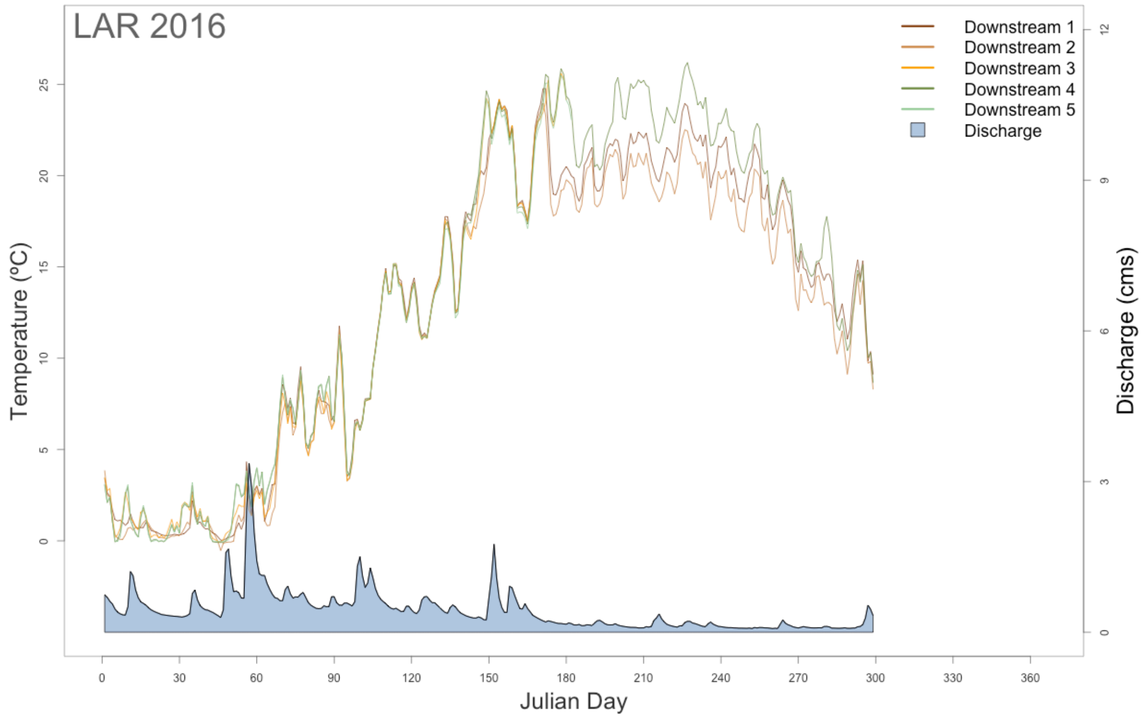
Mean daily temperature from the 4 loggers deployed upstream, within the impoundment, and downstream of Ipswich Mills Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



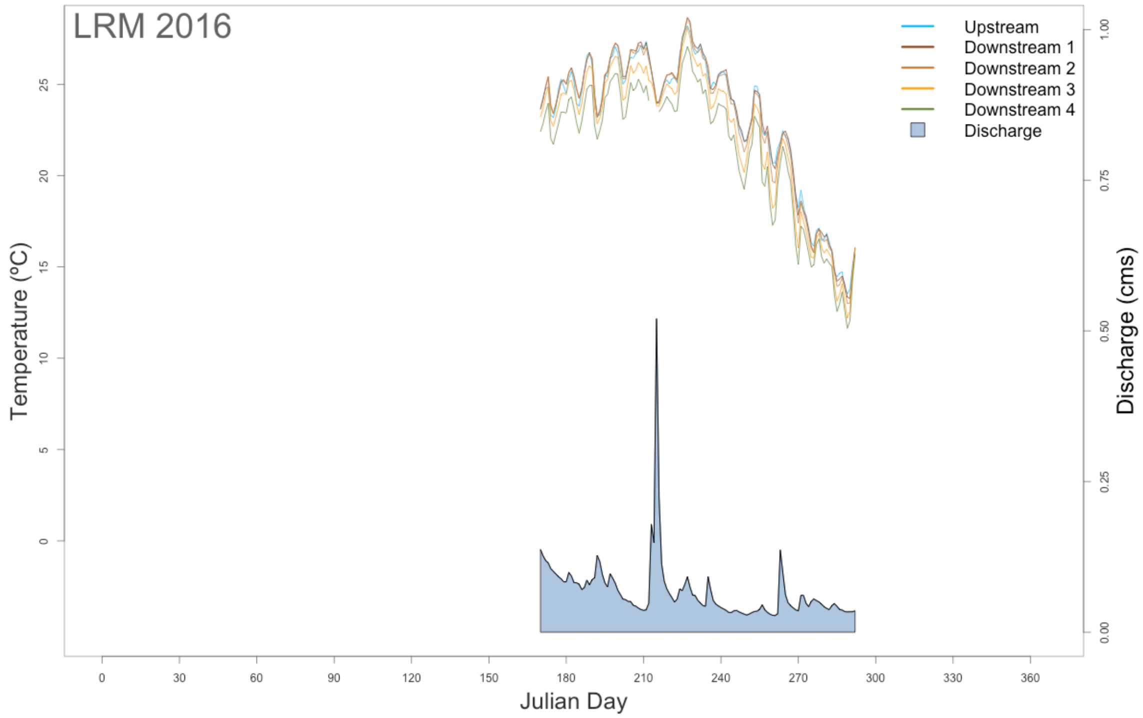
Mean daily temperature from the 5 loggers deployed downstream of Larrywaug / Stockbridge Bowl Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



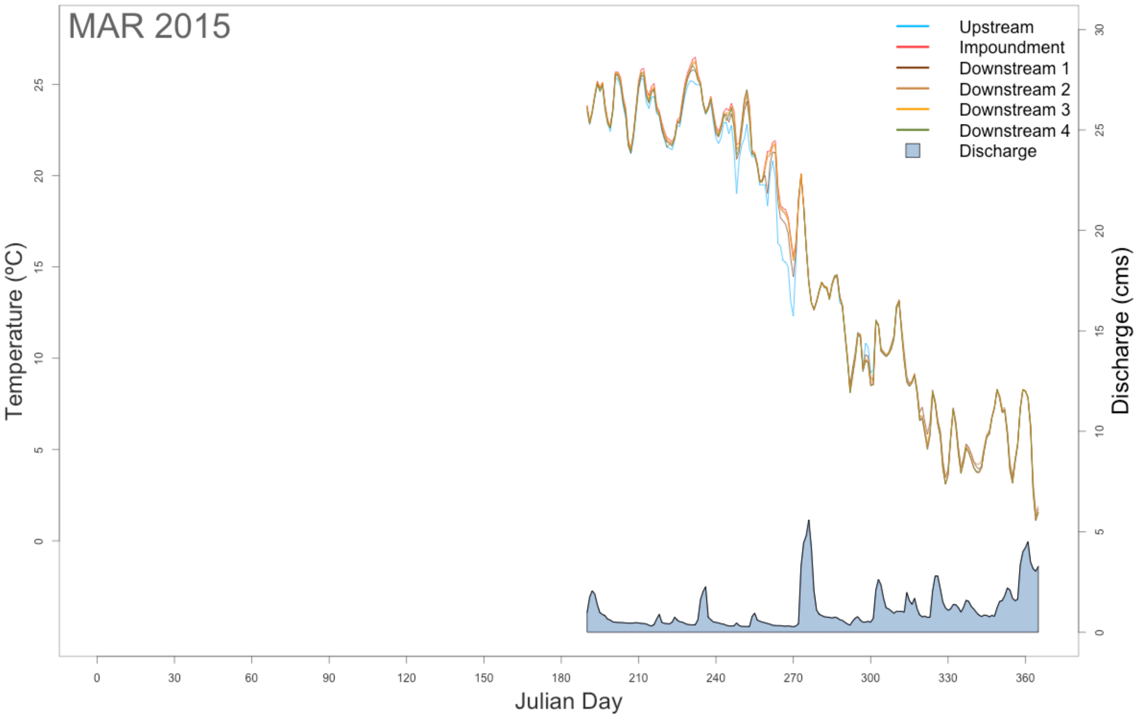
Mean daily temperature from the 5 loggers deployed downstream of Larrywaug / Stockbridge Bowl Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



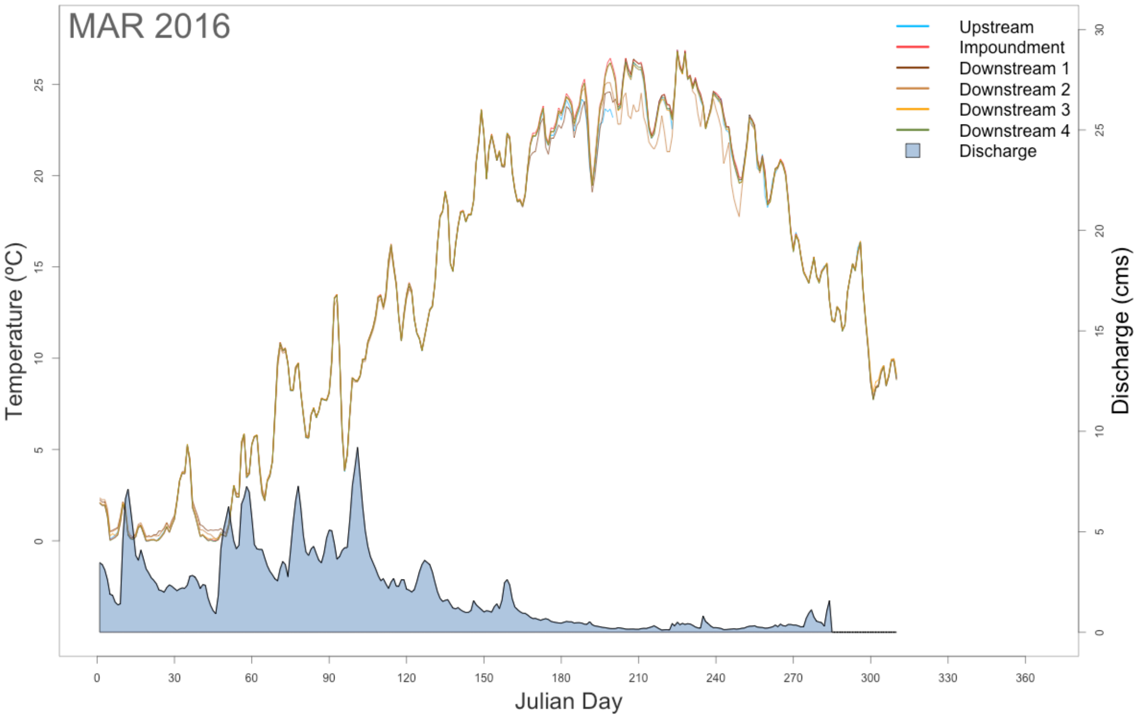
Mean daily temperature from the 5 loggers deployed downstream of Larrywaug / Stockbridge Bowl Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



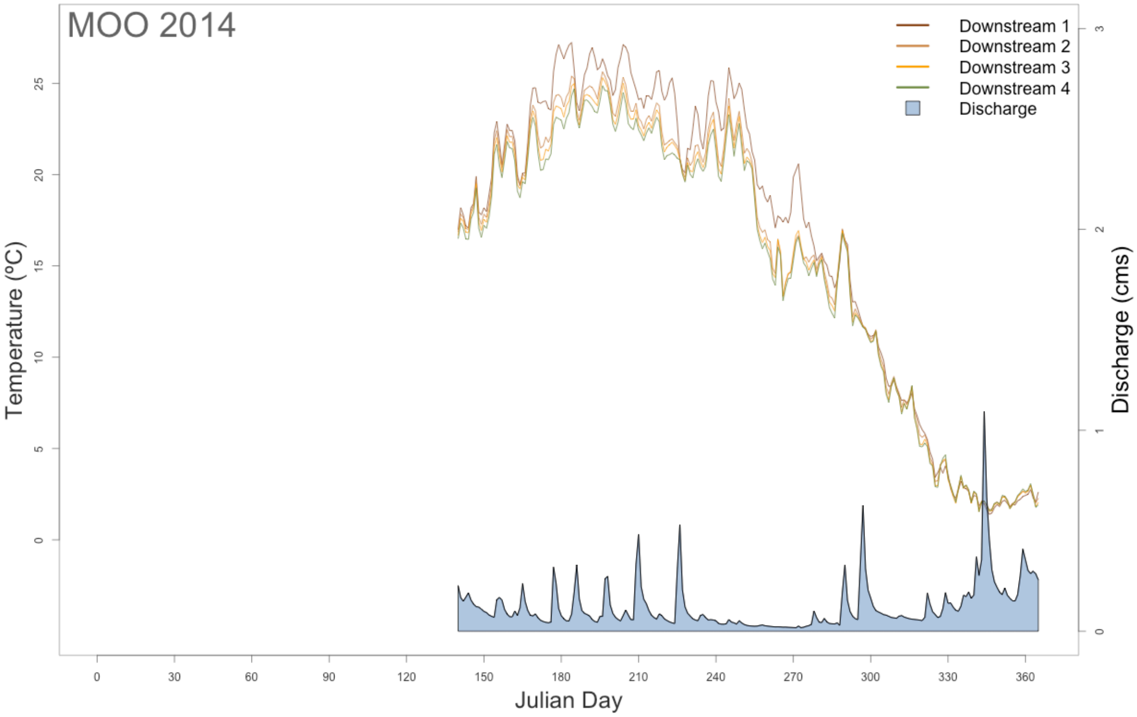
Mean daily temperature from the 5 loggers deployed upstream and downstream of Lower Roberts Meadow Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



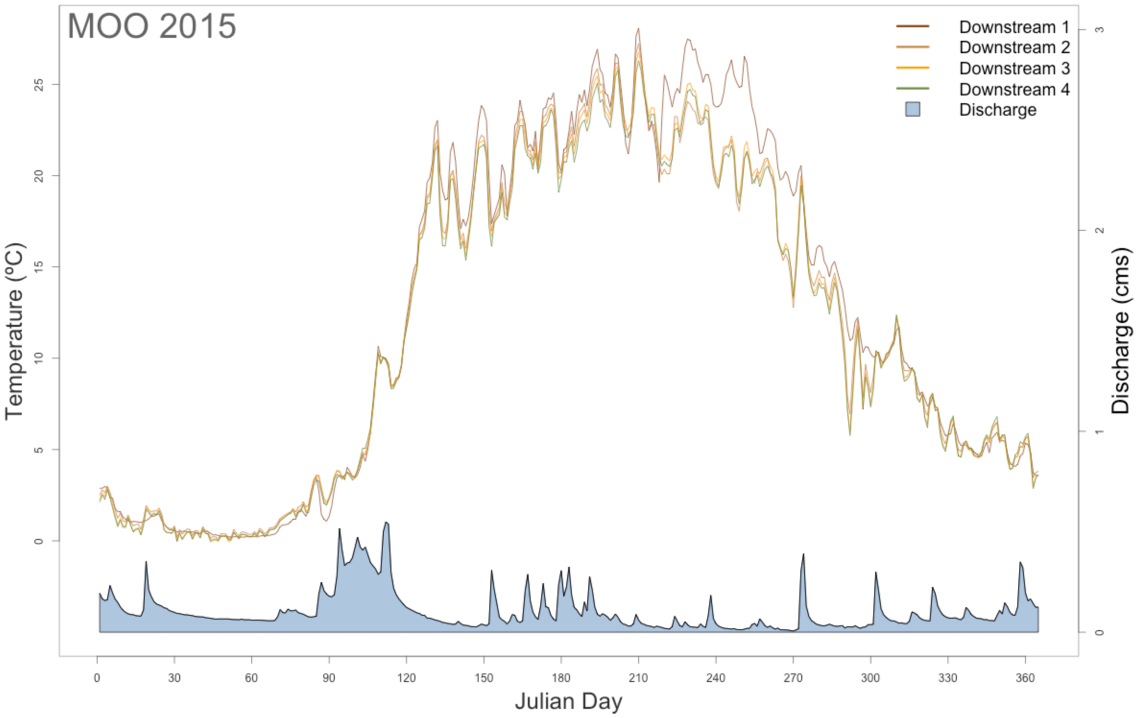
Mean daily temperature from the 6 loggers deployed upstream, within the impoundment, and downstream of Marland Place Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



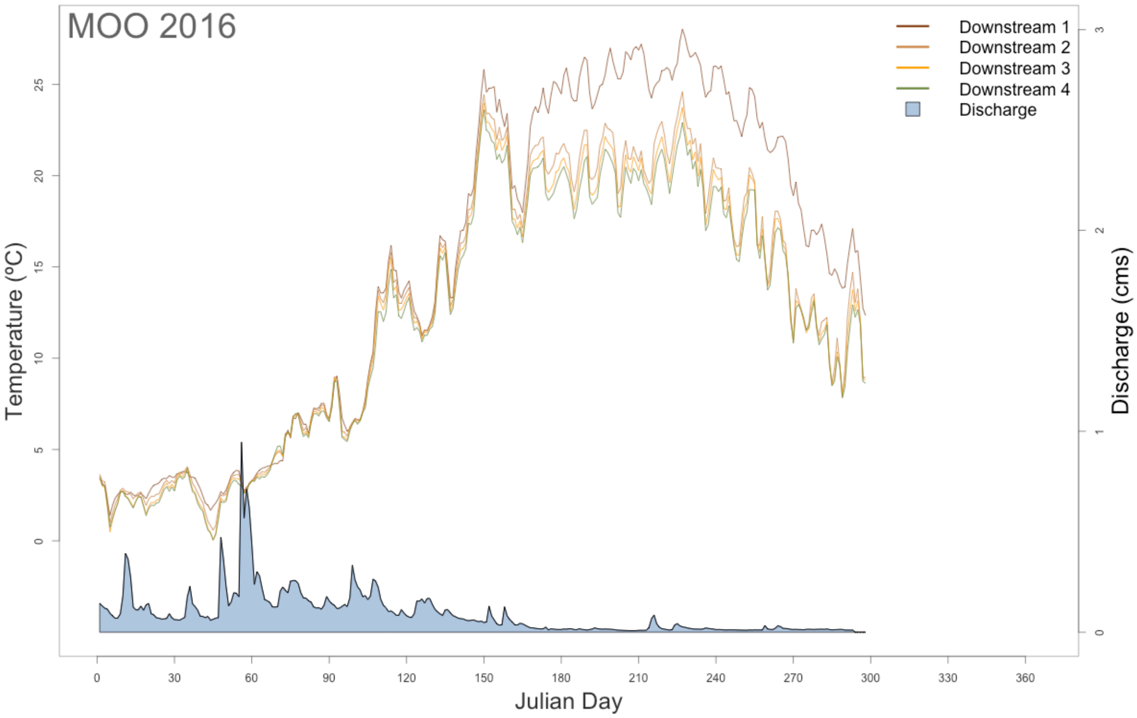
Mean daily temperature from the 6 loggers deployed upstream, within the impoundment, and downstream of Marland Place Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



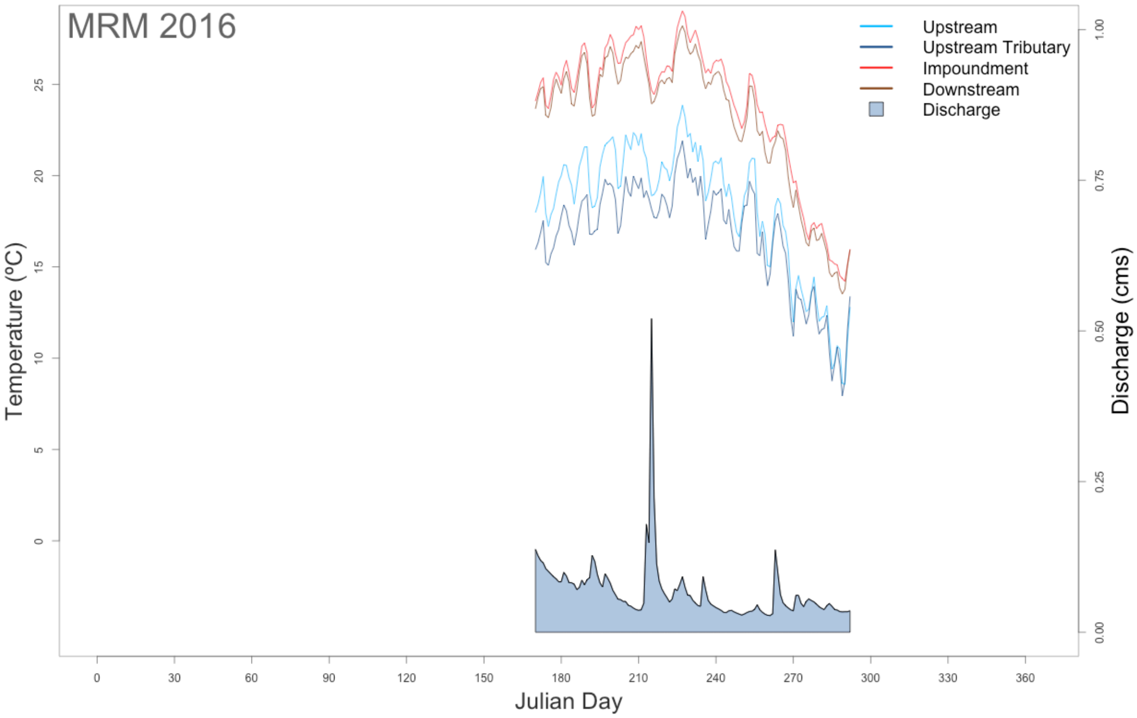
Mean daily temperature from the 4 loggers deployed downstream of Moose Meadow Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



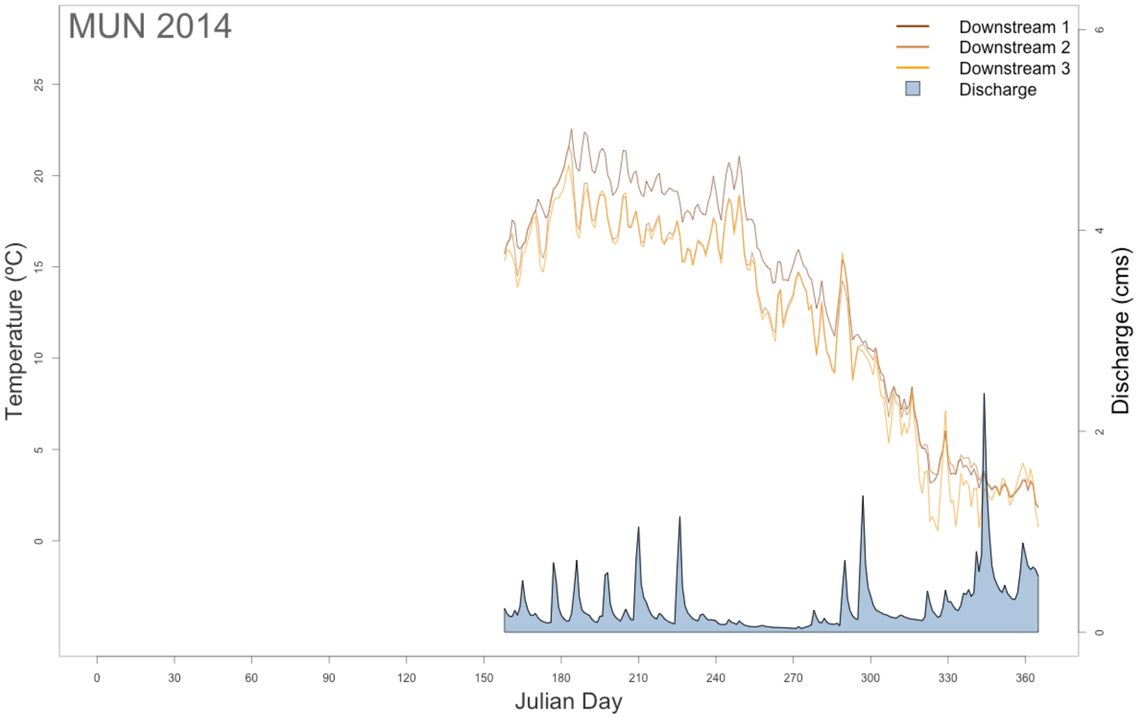
Mean daily temperature from the 4 loggers deployed downstream of Moose Meadow Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



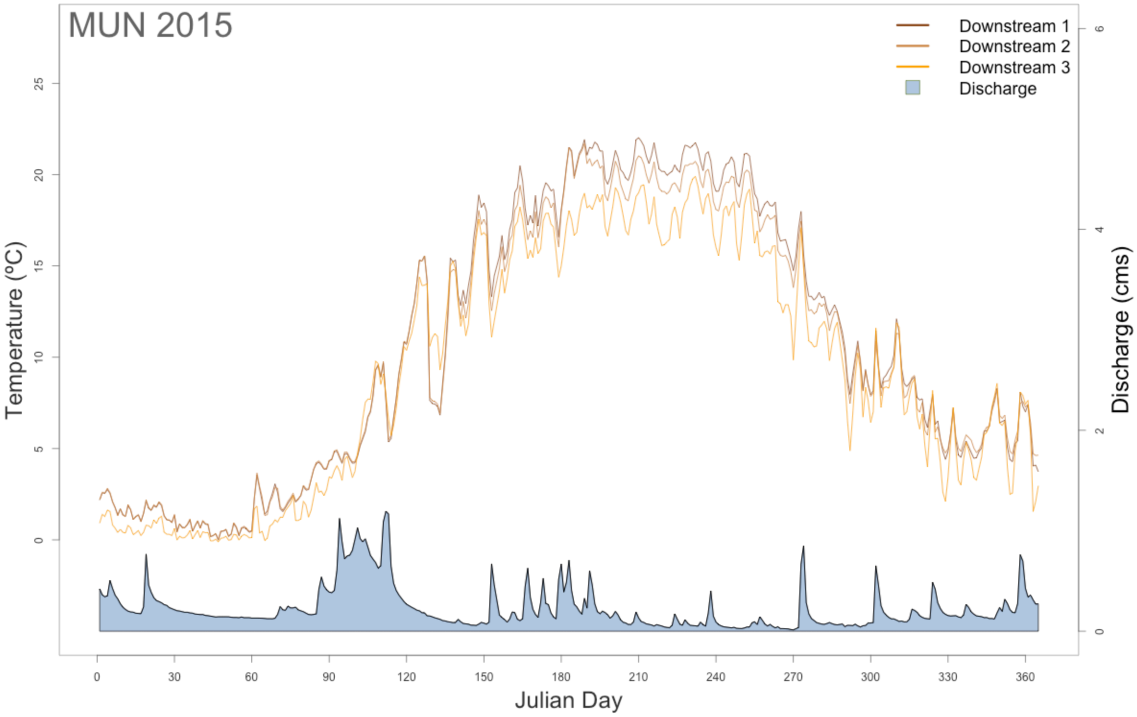
Mean daily temperature from the 4 loggers deployed downstream of Moose Meadow Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



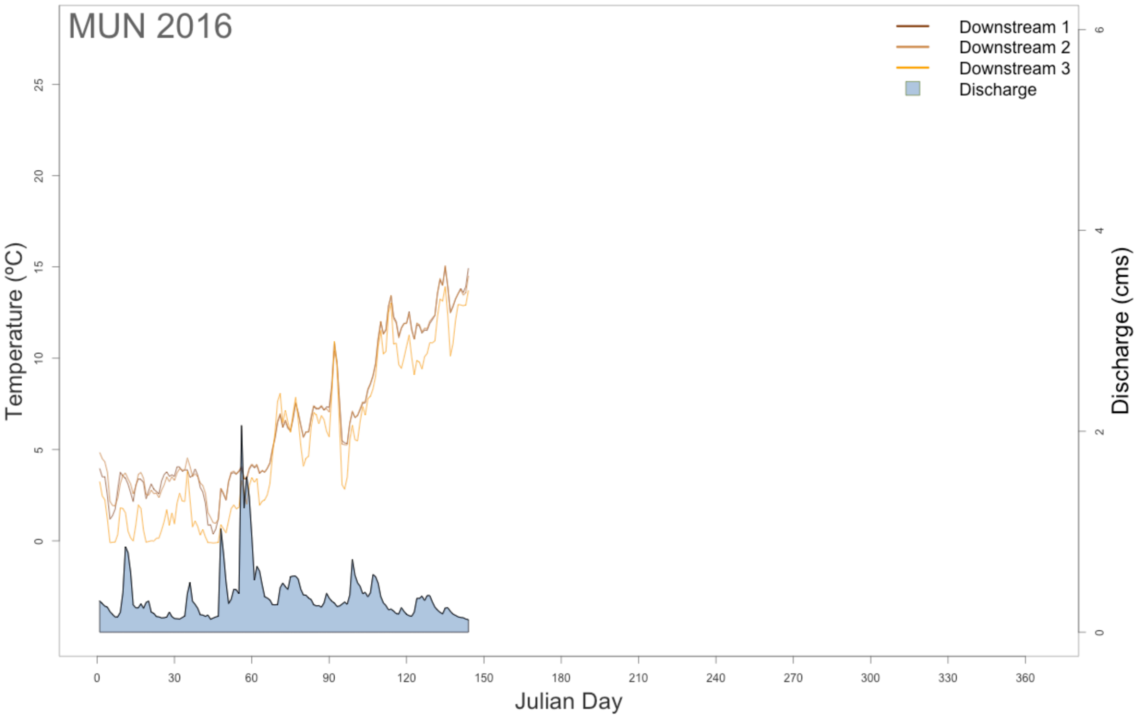
Mean daily temperature from the 4 loggers deployed upstream, within the impoundment, and downstream of Middle Roberts Meadow Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



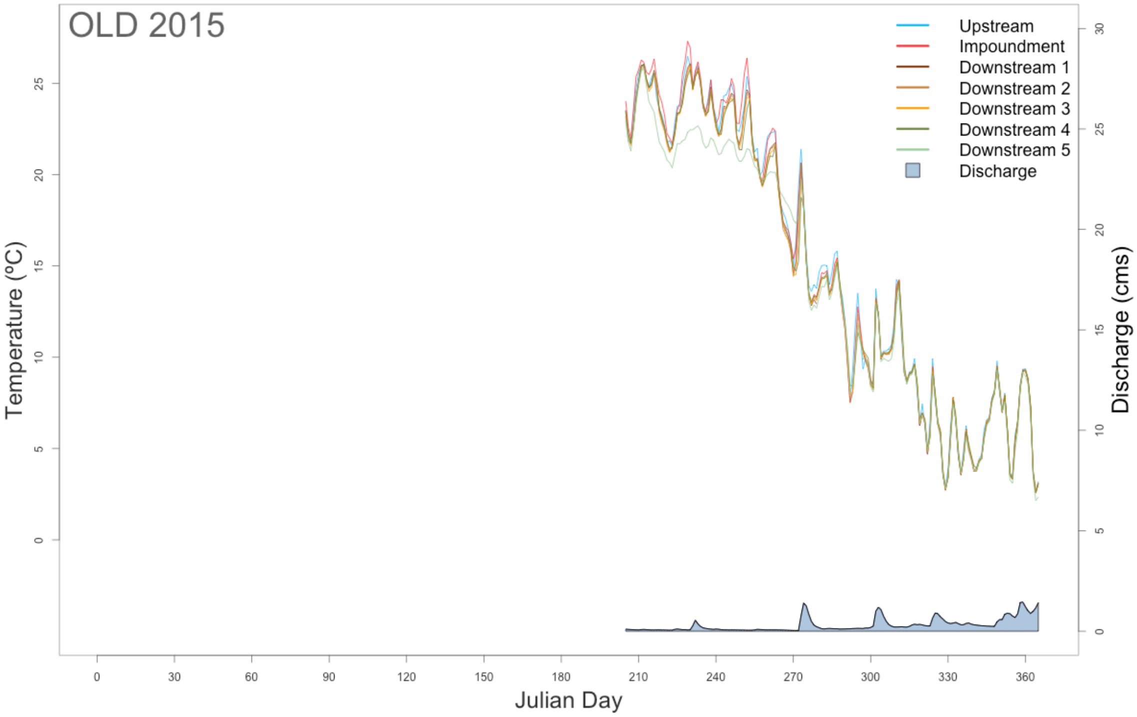
Mean daily temperature from the 3 loggers deployed downstream of Munn Brook Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



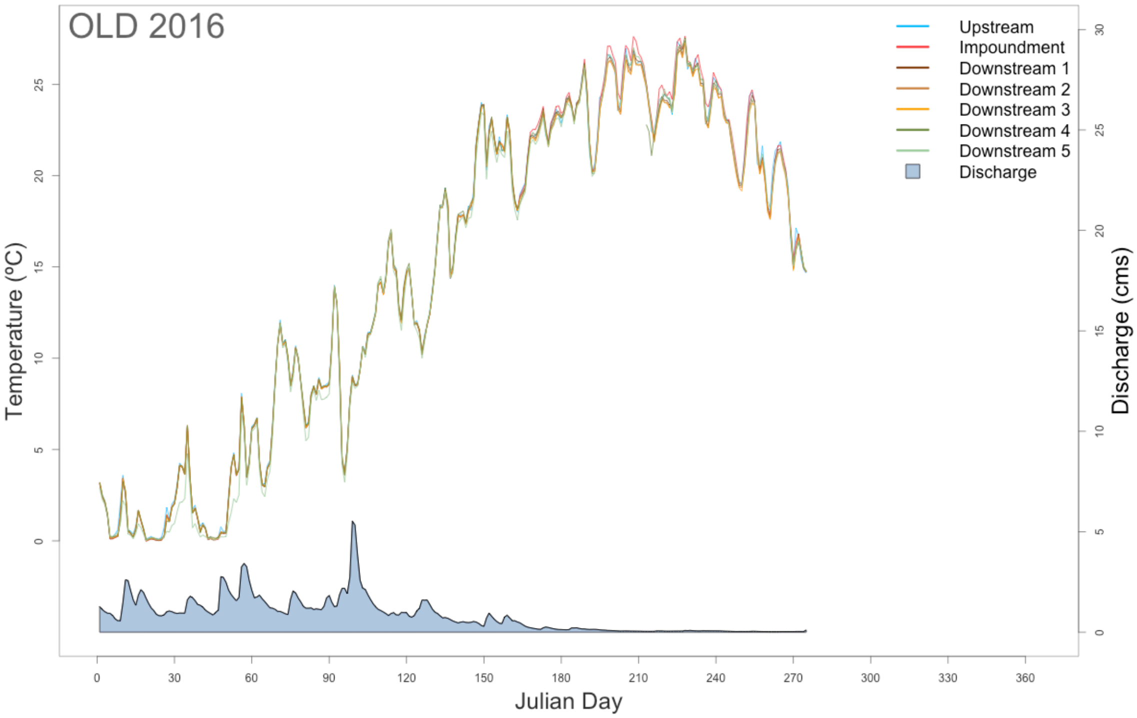
Mean daily temperature from the 3 loggers deployed downstream of Munn Brook Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



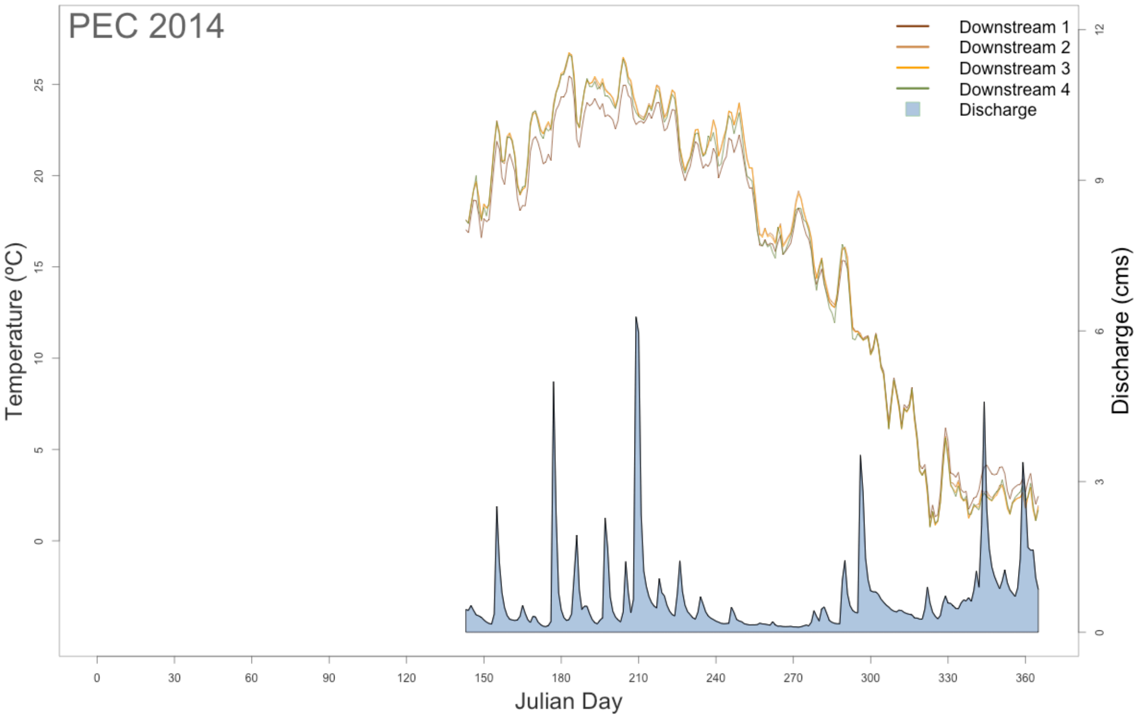
Mean daily temperature from the 3 loggers deployed downstream of Munn Brook Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



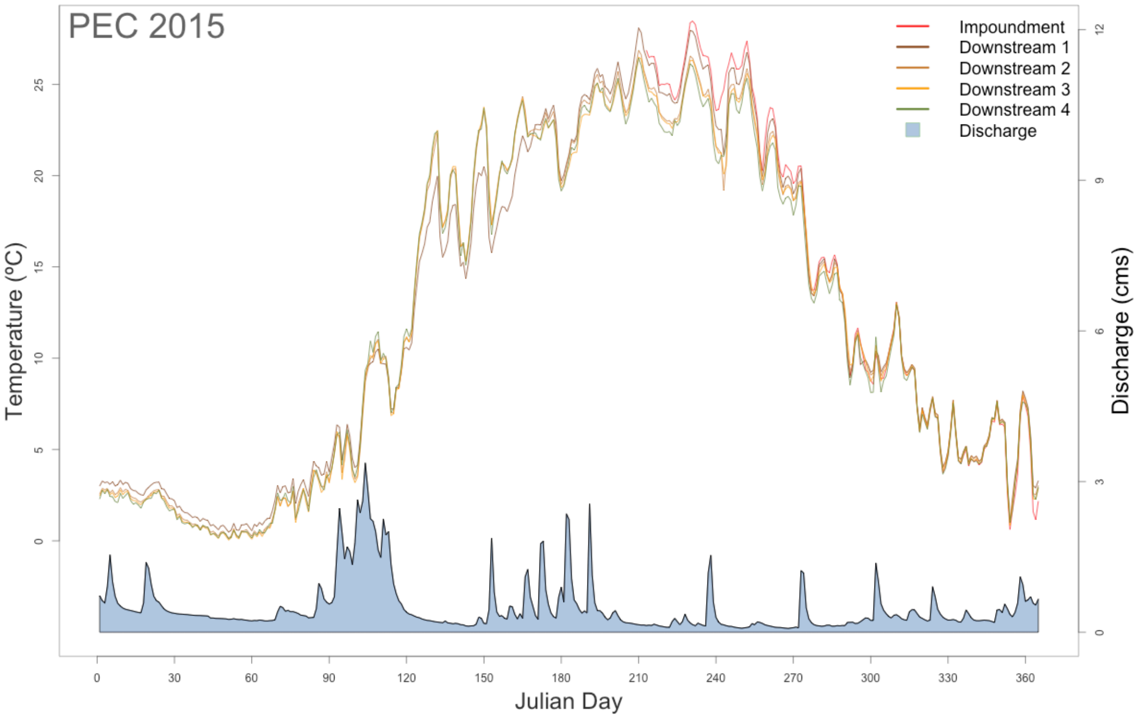
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Old Mill Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



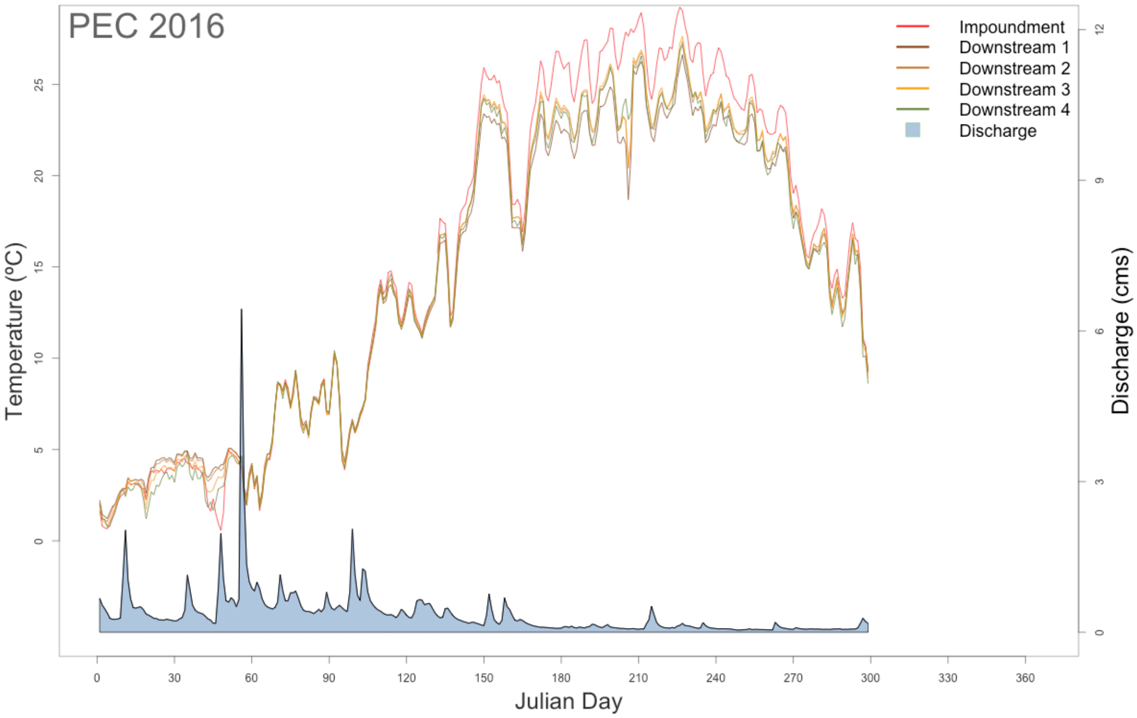
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Old Mill Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



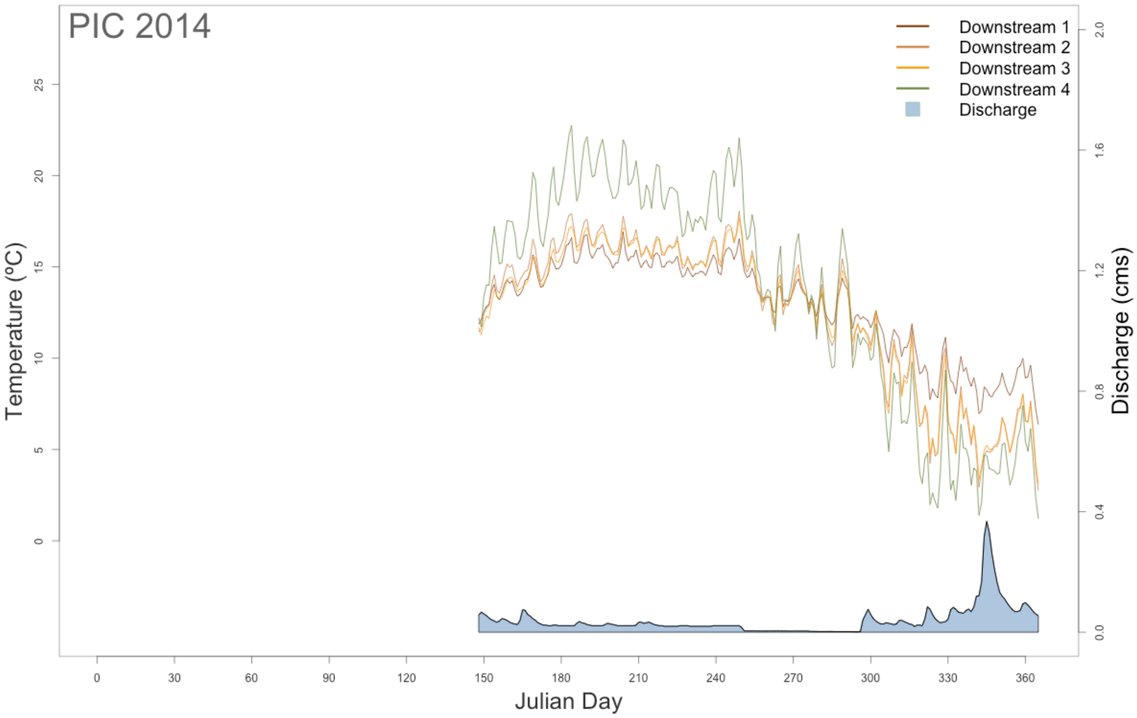
Mean daily temperature from the 4 loggers deployed downstream of Peck's Pond / Onota Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



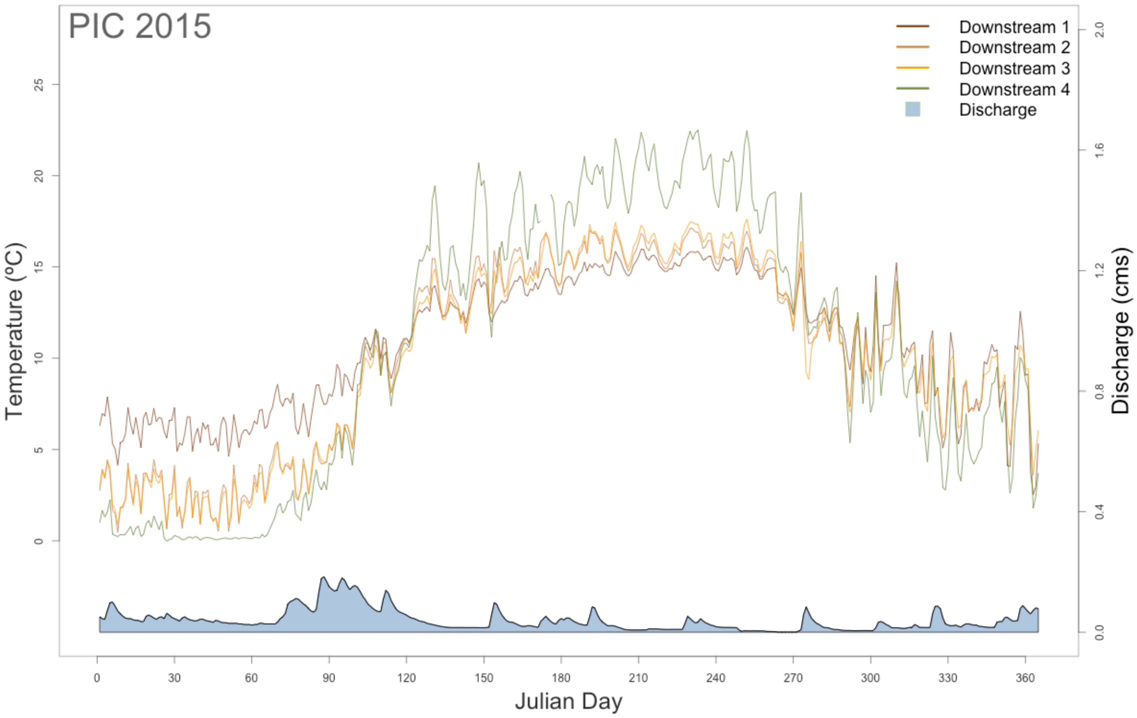
Mean daily temperature from the 5 loggers deployed within the impoundment and downstream of Peck's Pond / Onota Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



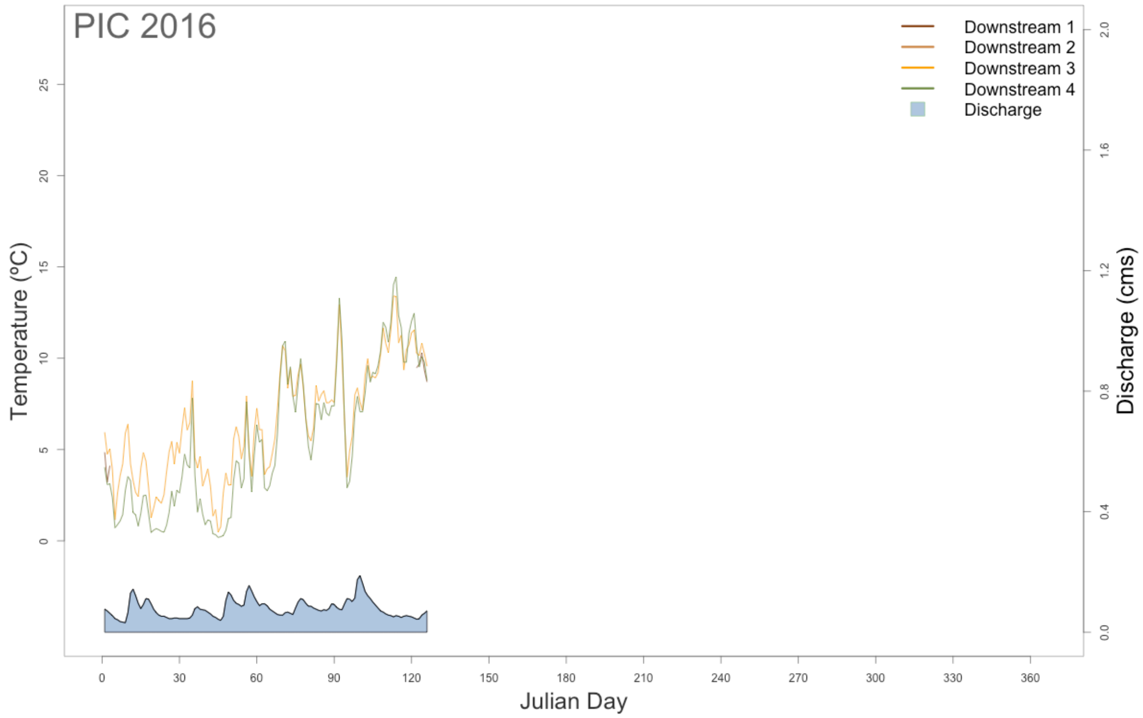
Mean daily temperature from the 5 loggers deployed within the impoundment and downstream of Peck's Pond / Onota Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



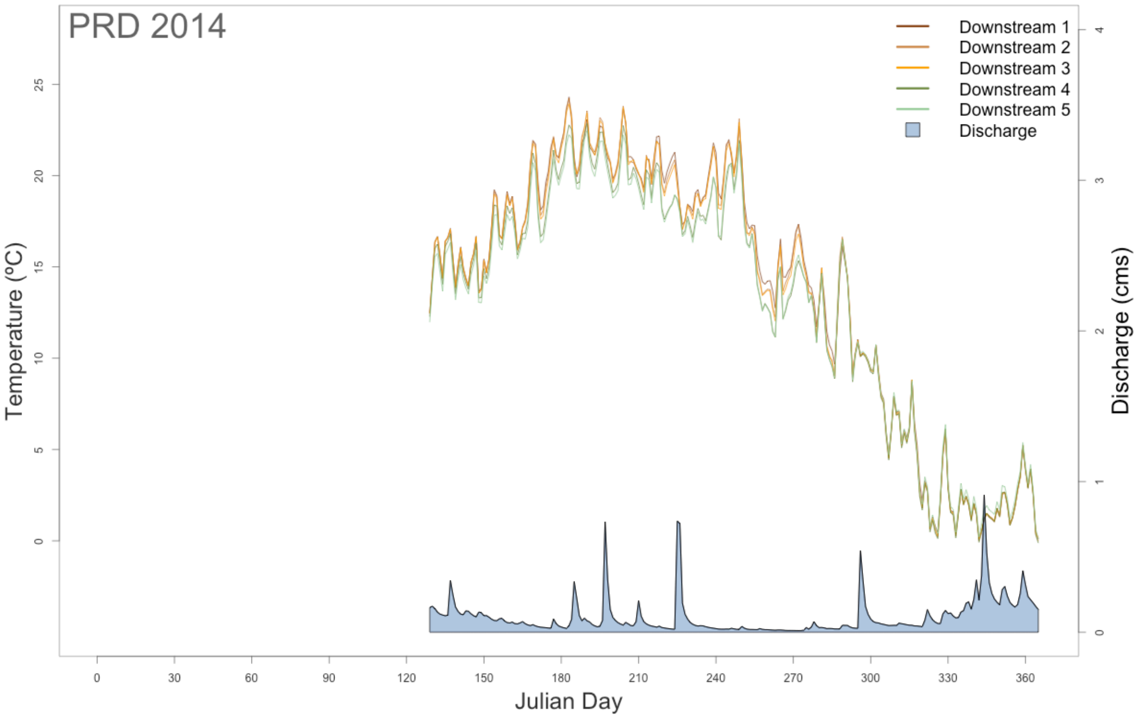
Mean daily temperature from the 4 loggers deployed downstream of Piccadilly Brook Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



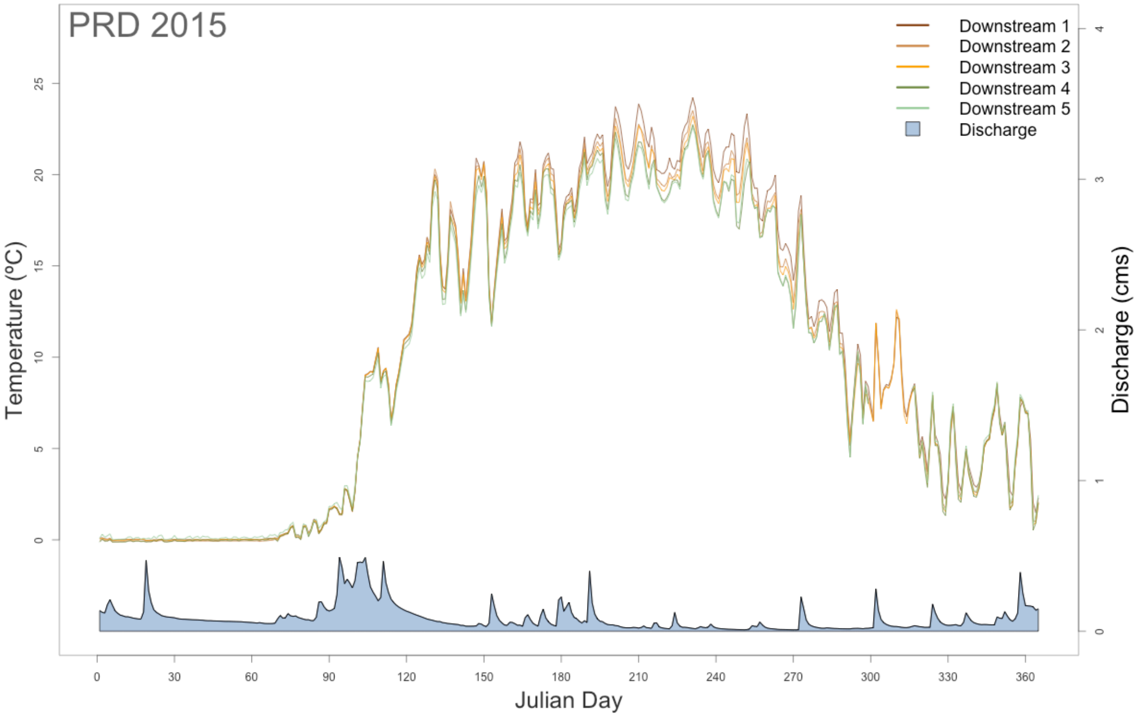
Mean daily temperature from the 4 loggers deployed downstream of Piccadilly Brook Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



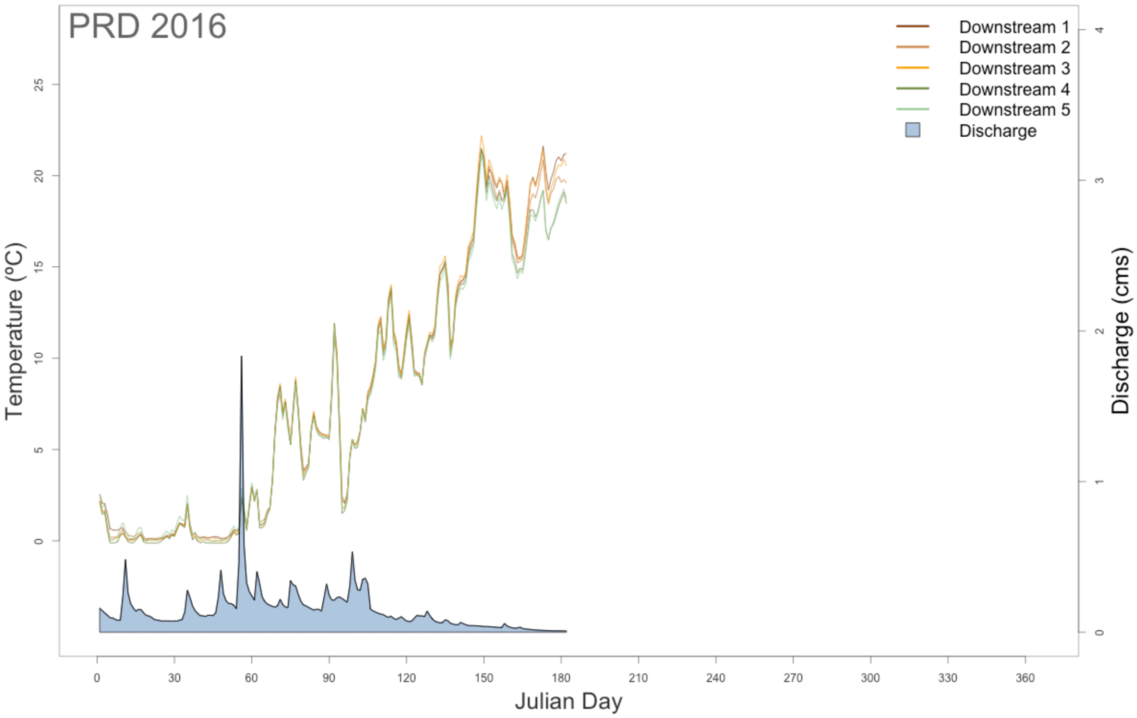
Mean daily temperature from the 4 loggers deployed downstream of Piccadilly Brook Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



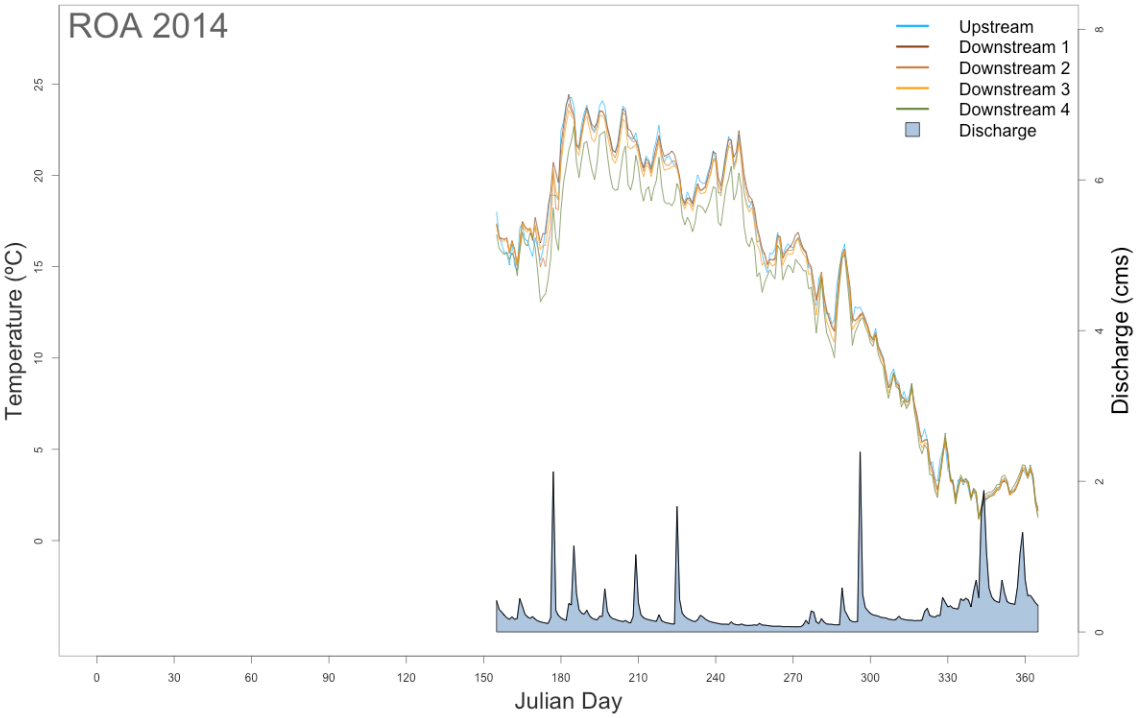
Mean daily temperature from the 5 loggers deployed downstream of Prescott Road 17 beaver dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



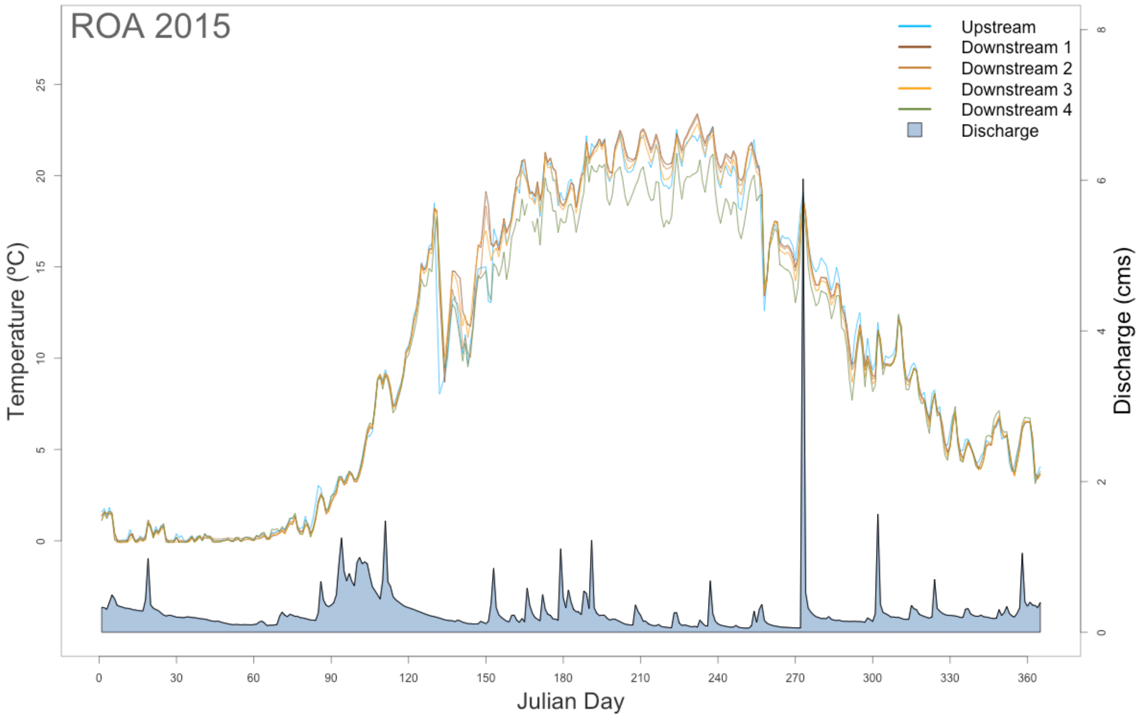
Mean daily temperature from the 5 loggers deployed downstream of Prescott Road 17 beaver dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



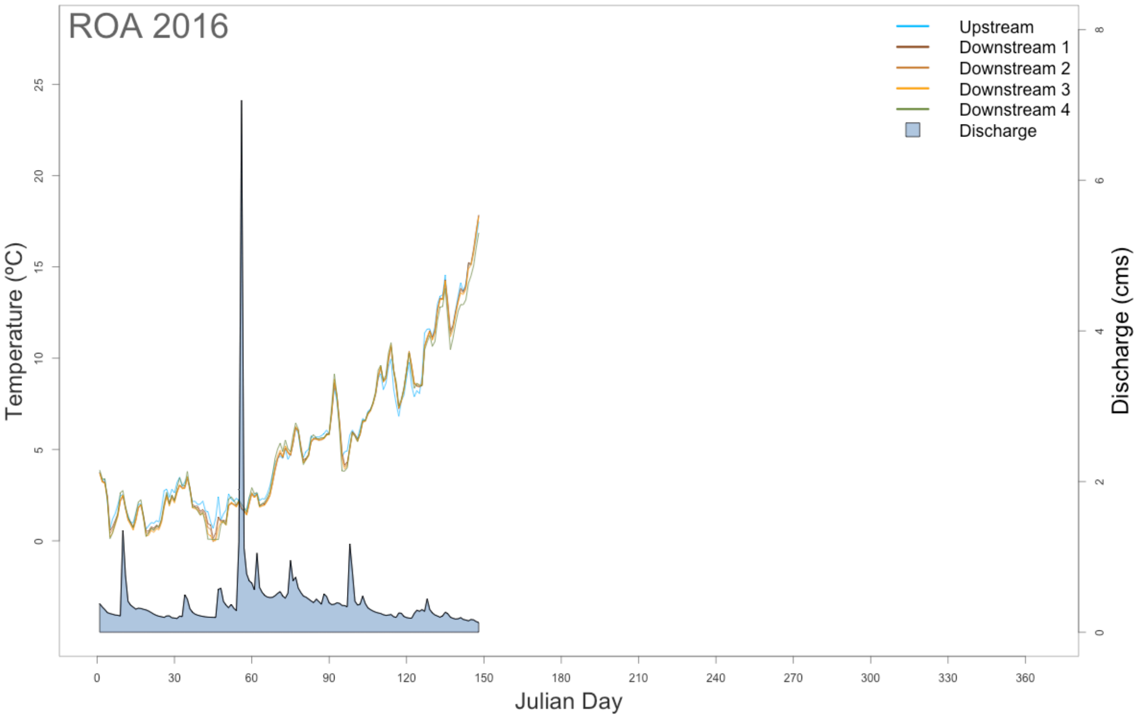
Mean daily temperature from the 5 loggers deployed downstream of Prescott Road 17 beaver dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



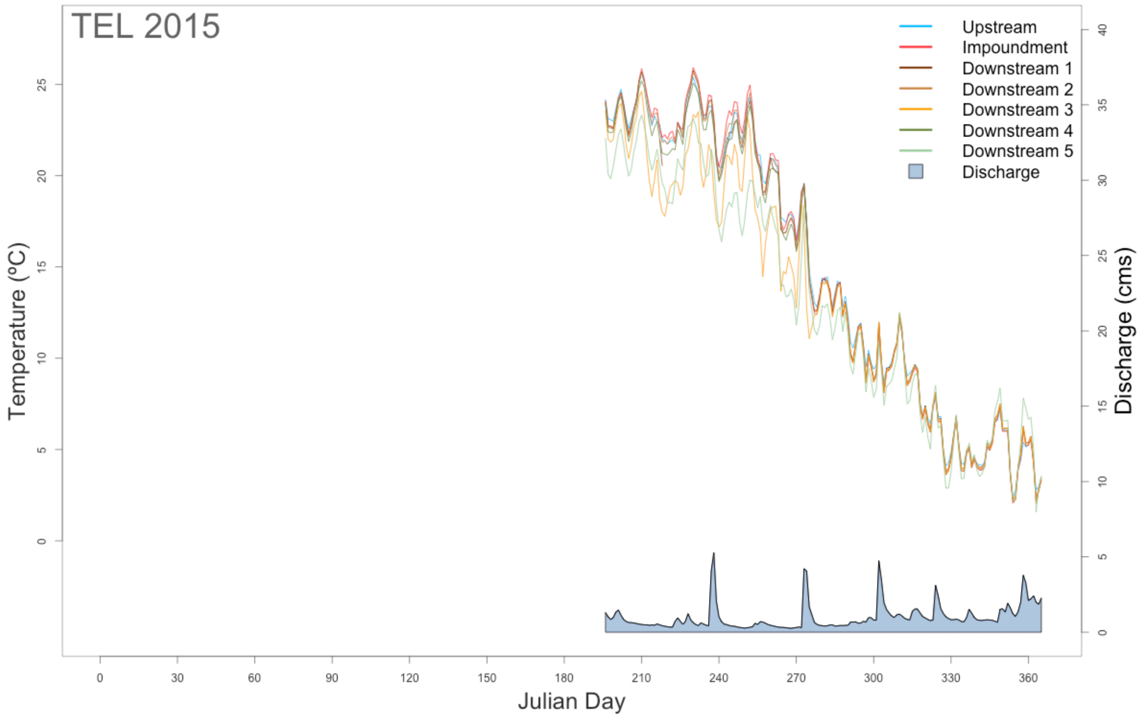
Mean daily temperature from the 5 loggers deployed upstream and downstream of Roaring Brook Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



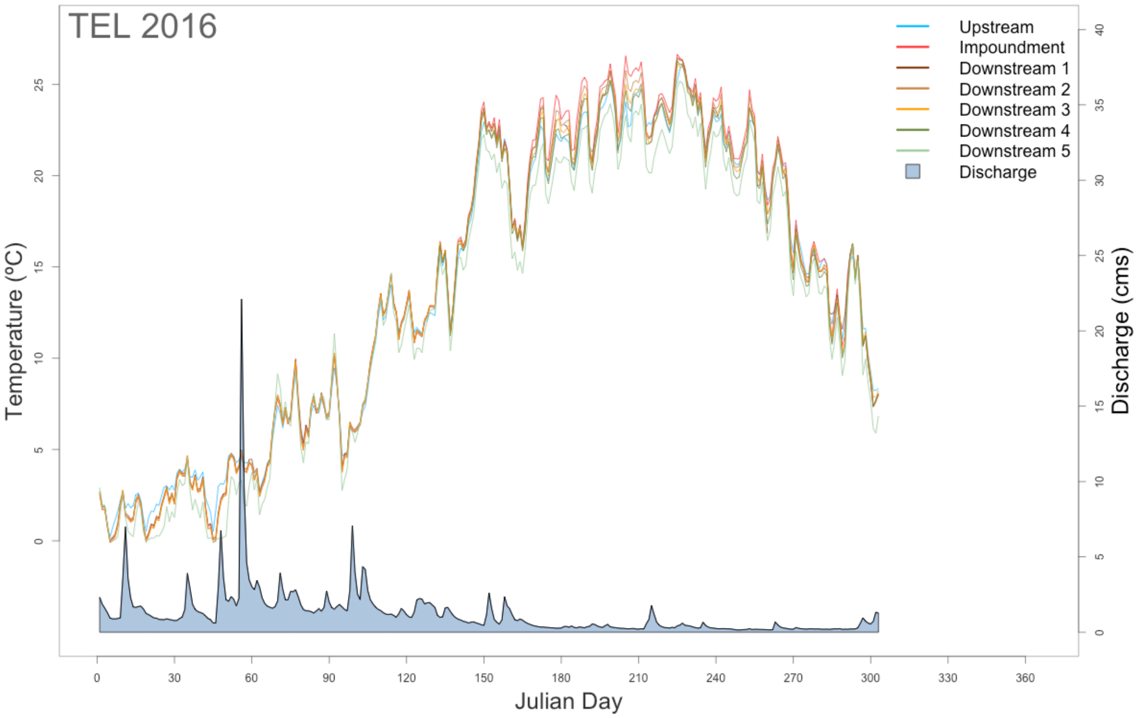
Mean daily temperature from the 5 loggers deployed upstream and downstream of Roaring Brook Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



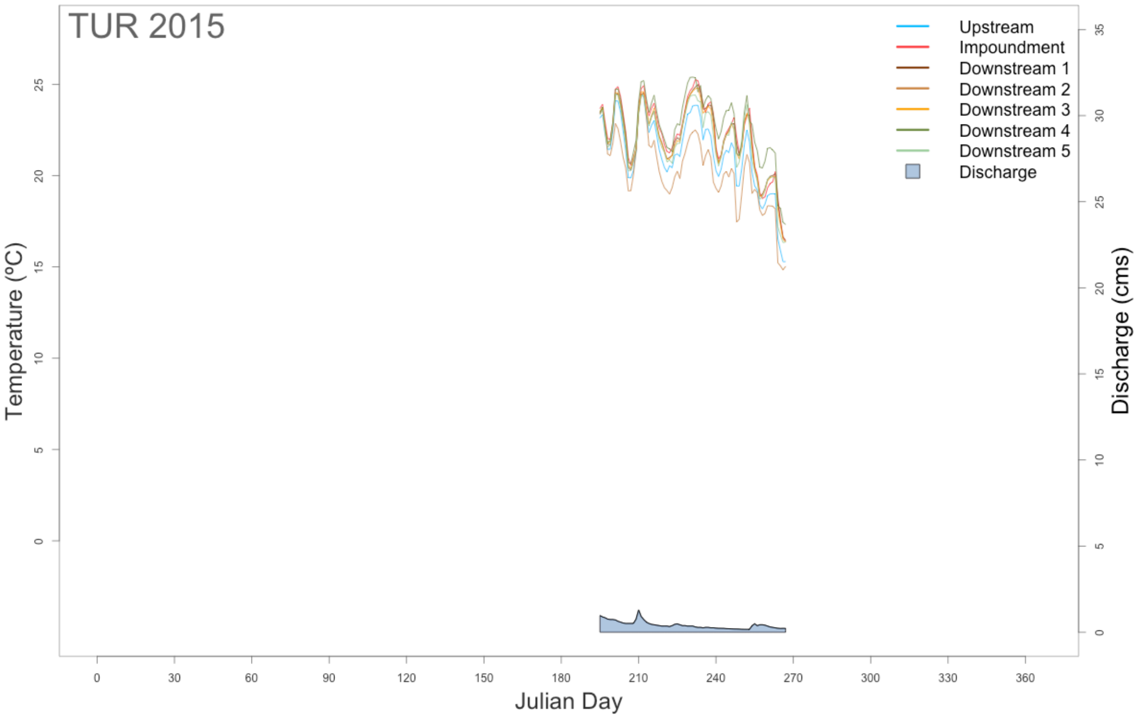
Mean daily temperature from the 5 loggers deployed upstream and downstream of Roaring Brook Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



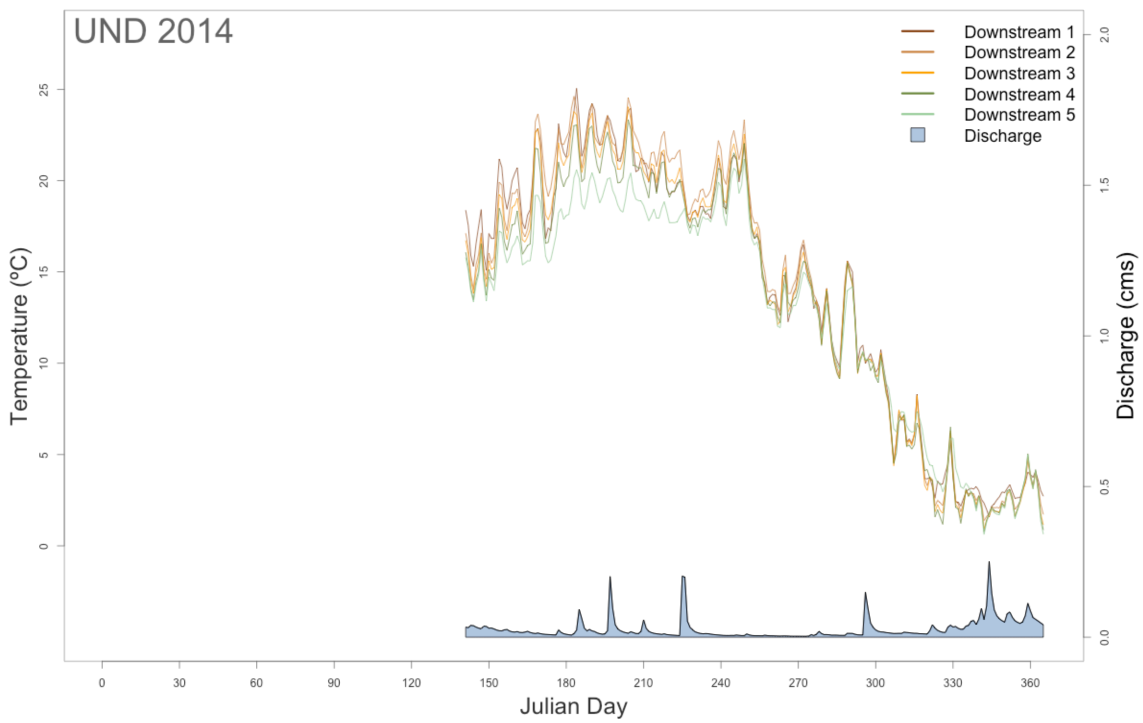
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Tel-Electric Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



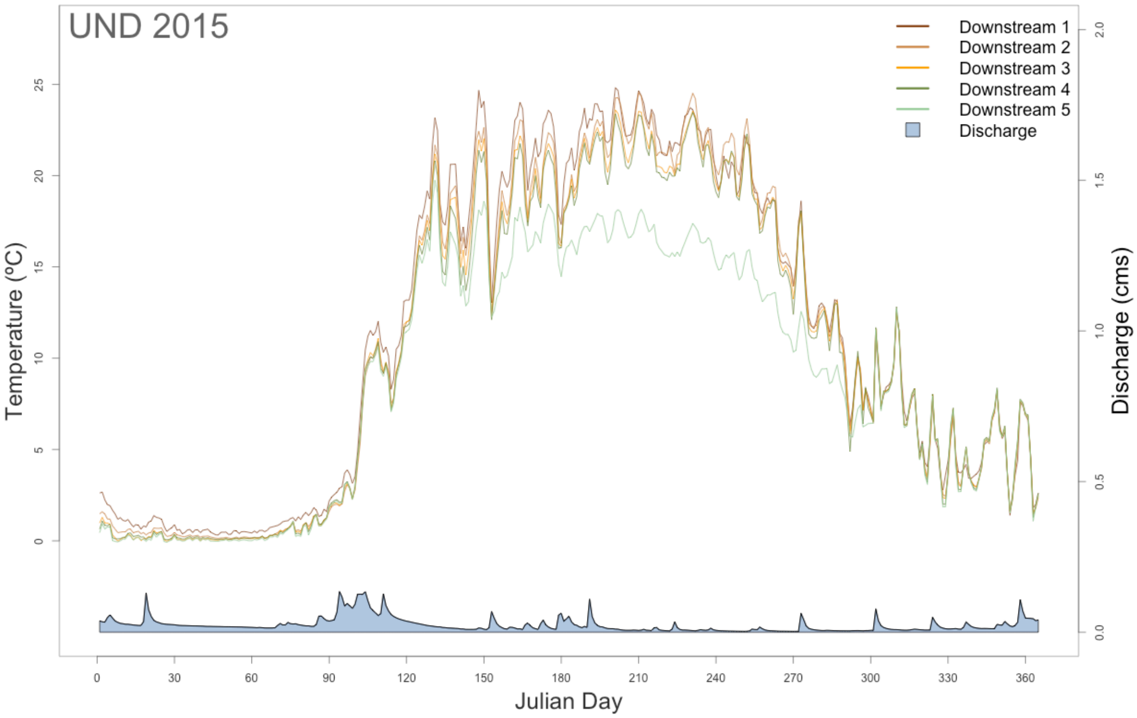
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Tel-Electric Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



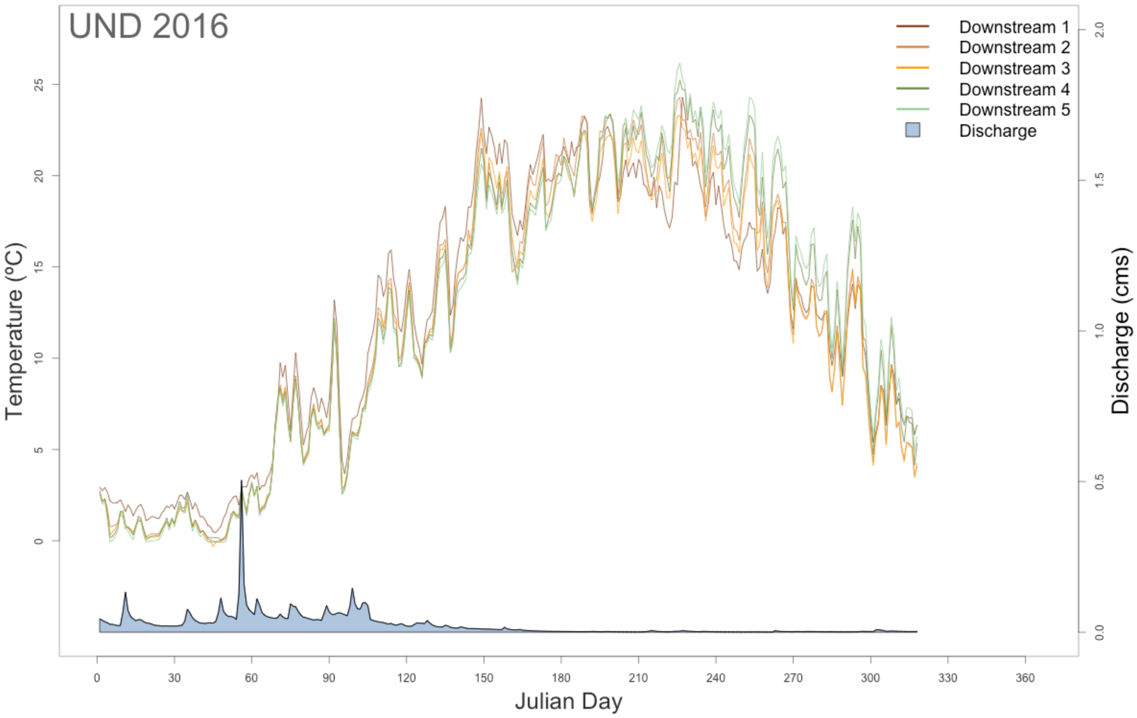
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Turner Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



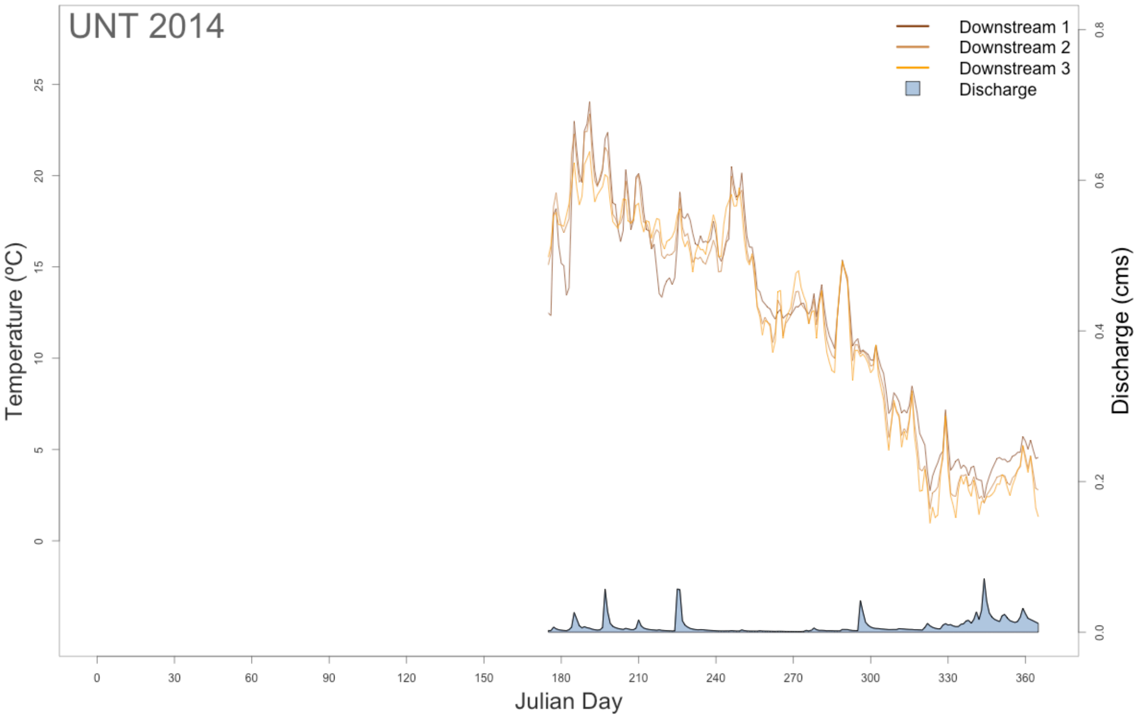
Mean daily temperature from the 5 loggers deployed downstream of Underhill Brook beaver dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



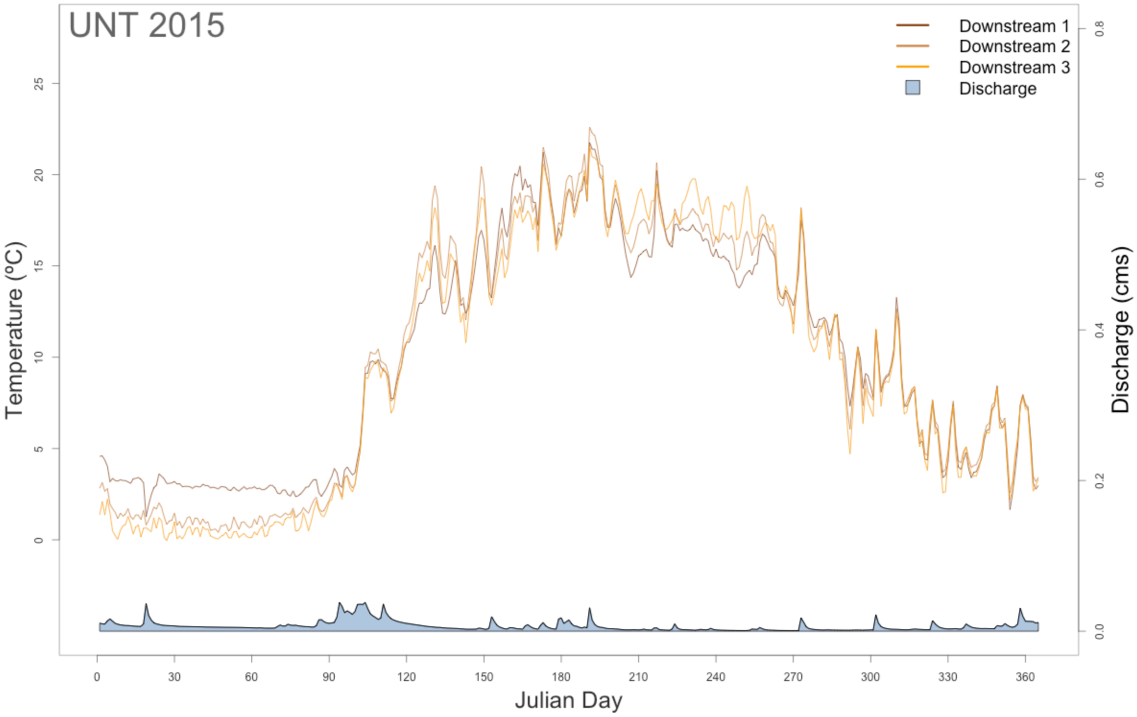
Mean daily temperature from the 5 loggers deployed downstream of Underhill Brook beaver dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



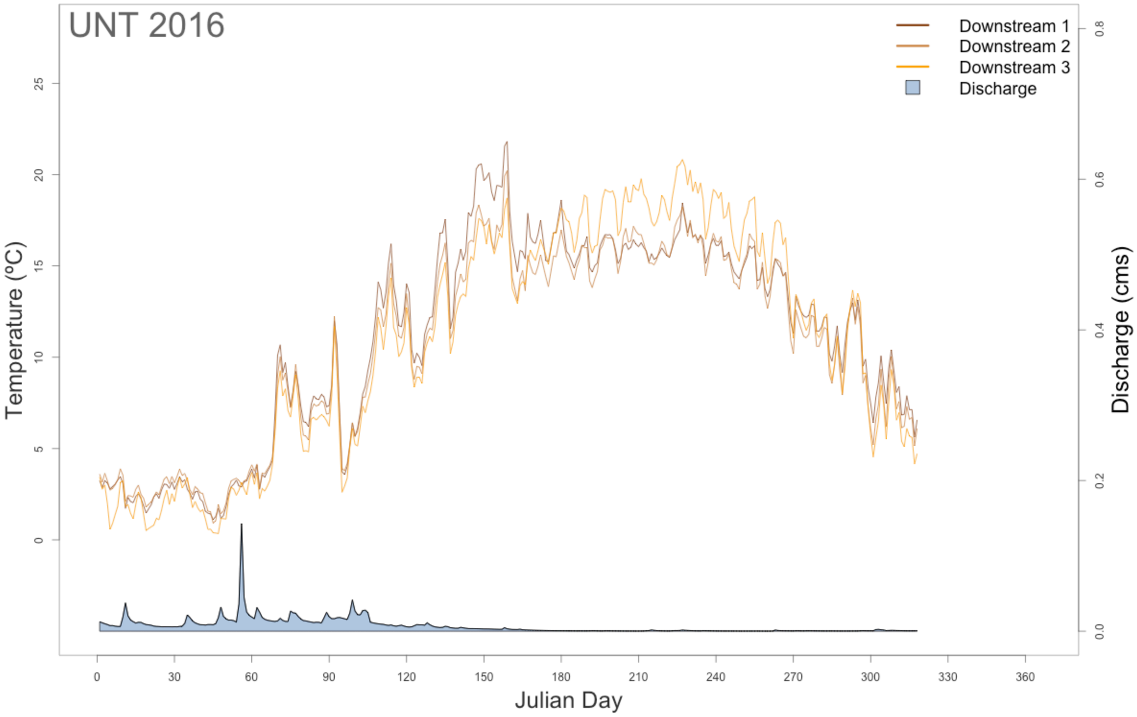
Mean daily temperature from the 5 loggers deployed downstream of Underhill Brook beaver dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



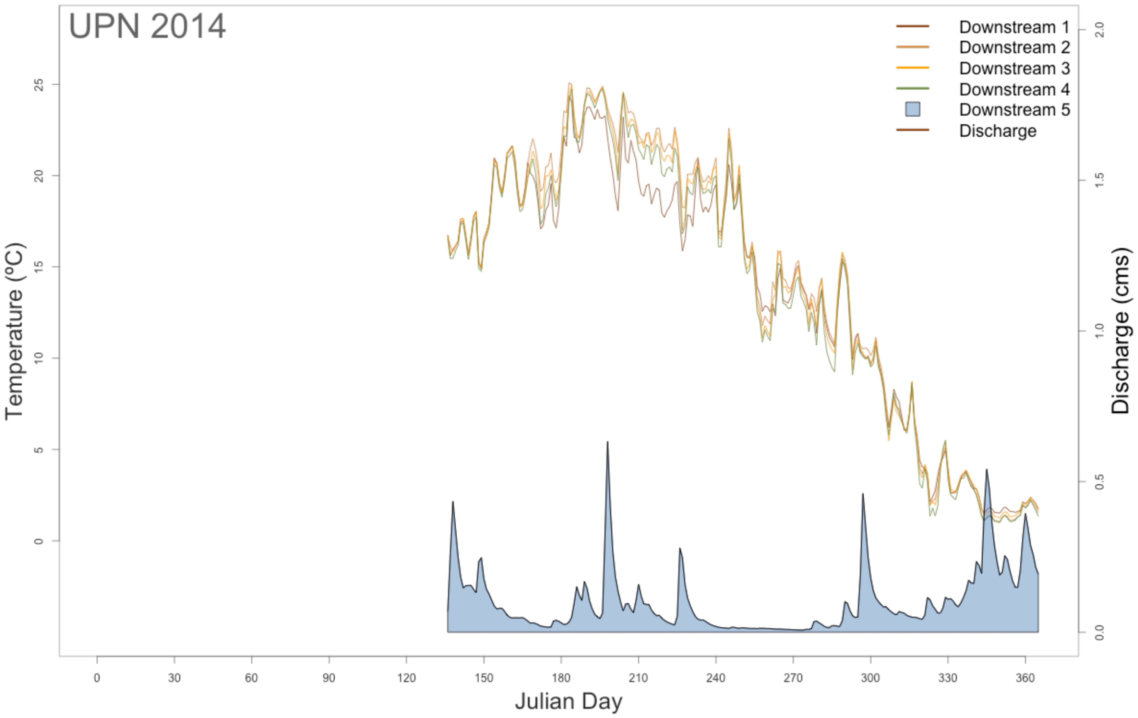
Mean daily temperature from the 3 loggers deployed downstream of Underhill Brook Tributary beaver dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



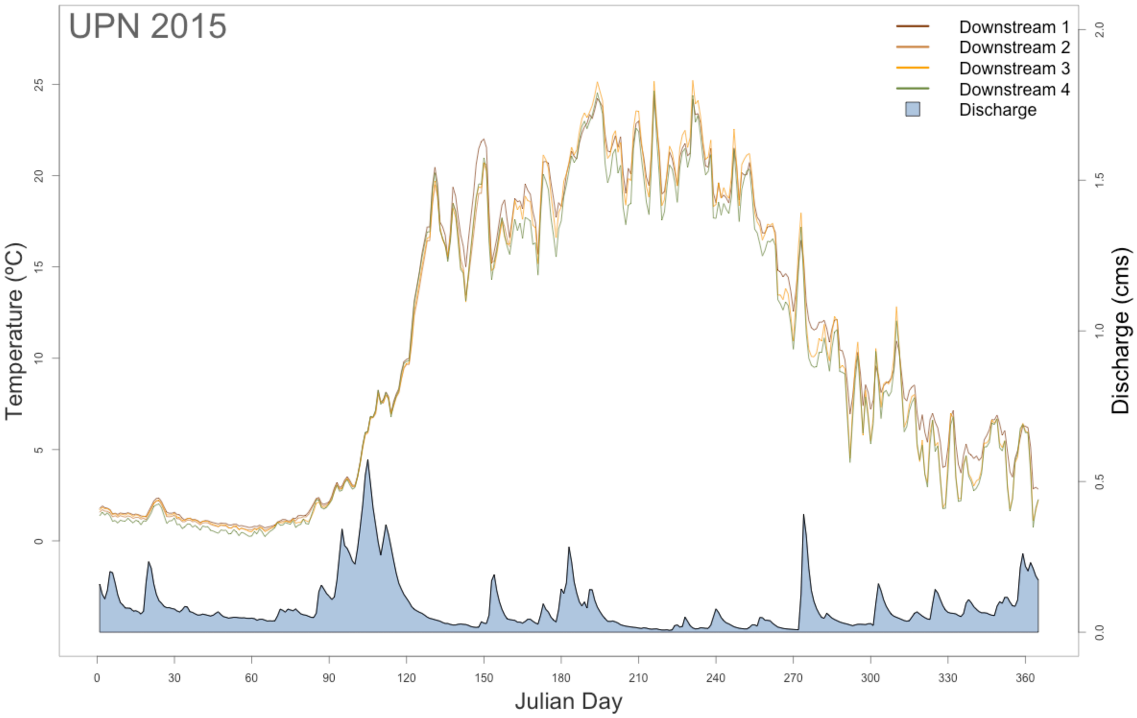
Mean daily temperature from the 3 loggers deployed downstream of Underhill Brook Tributary beaver dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



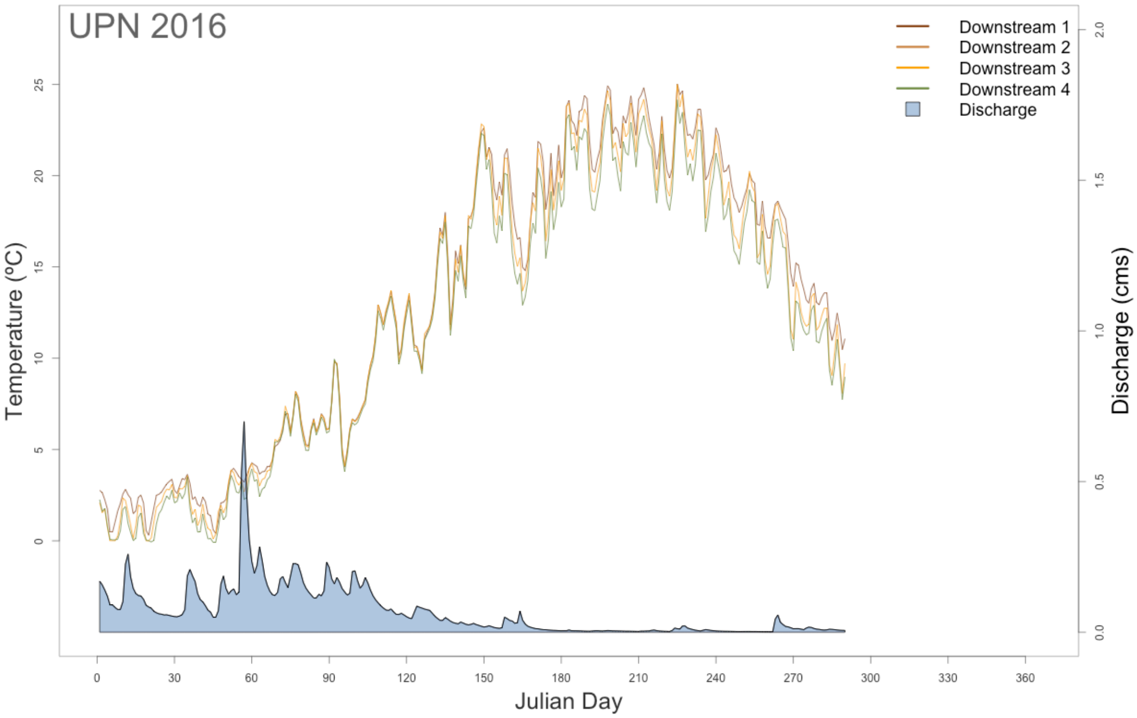
Mean daily temperature from the 3 loggers deployed downstream of Underhill Brook Tributary beaver dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



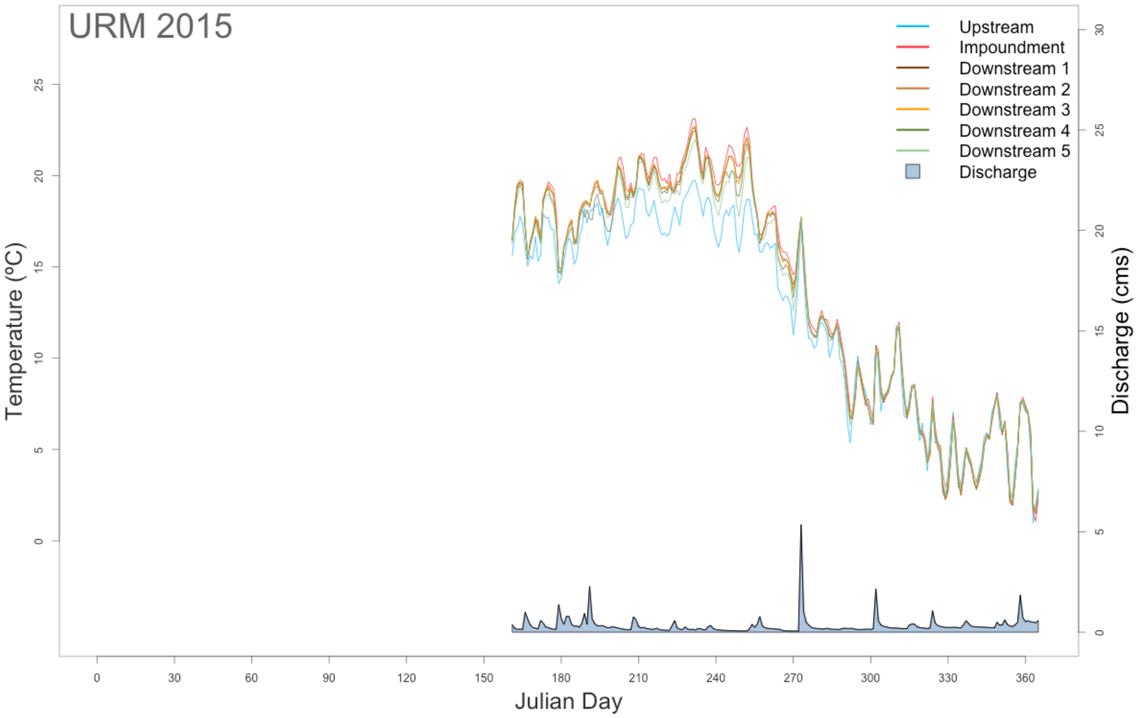
Mean daily temperature from the 4 loggers deployed downstream of Upper Naukeag Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



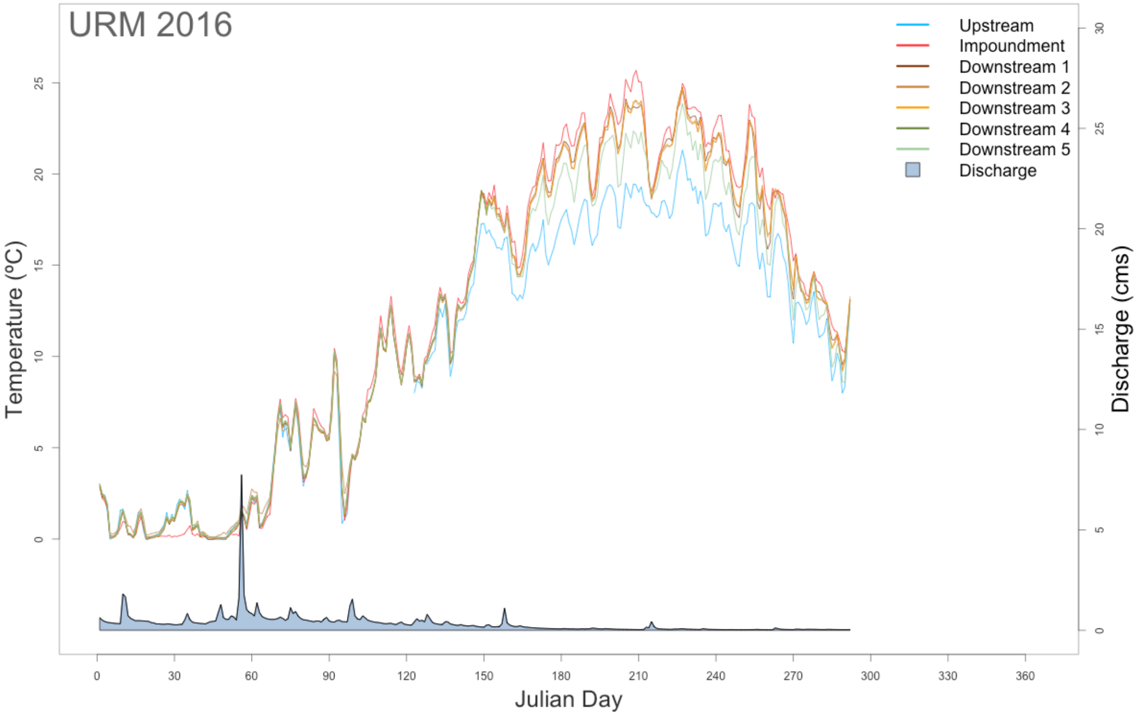
Mean daily temperature from the 4 loggers deployed downstream of Upper Naukeag Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



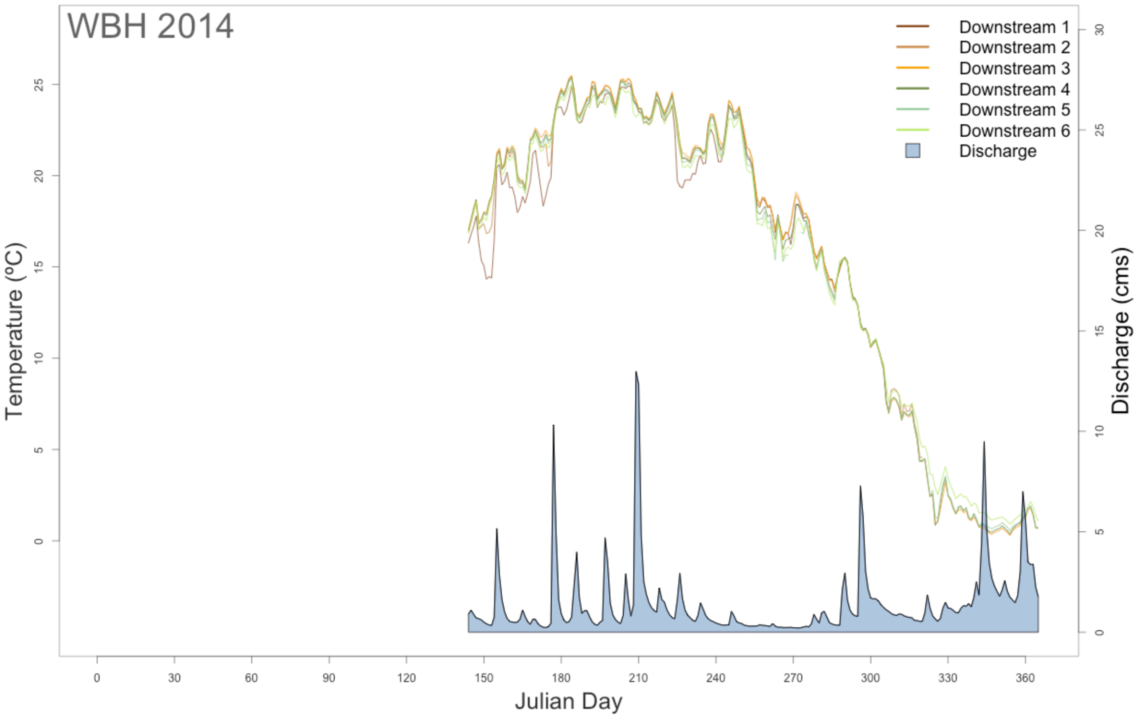
Mean daily temperature from the 4 loggers deployed downstream of Upper Naukeag Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



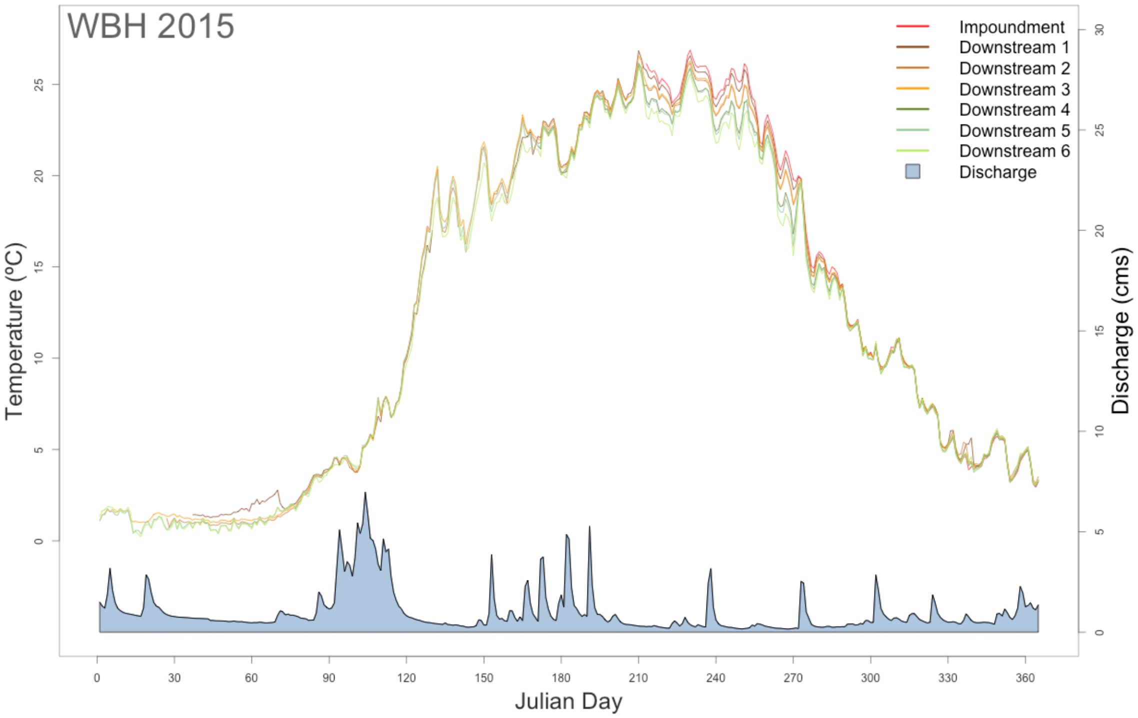
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Upper Roberts Meadow Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



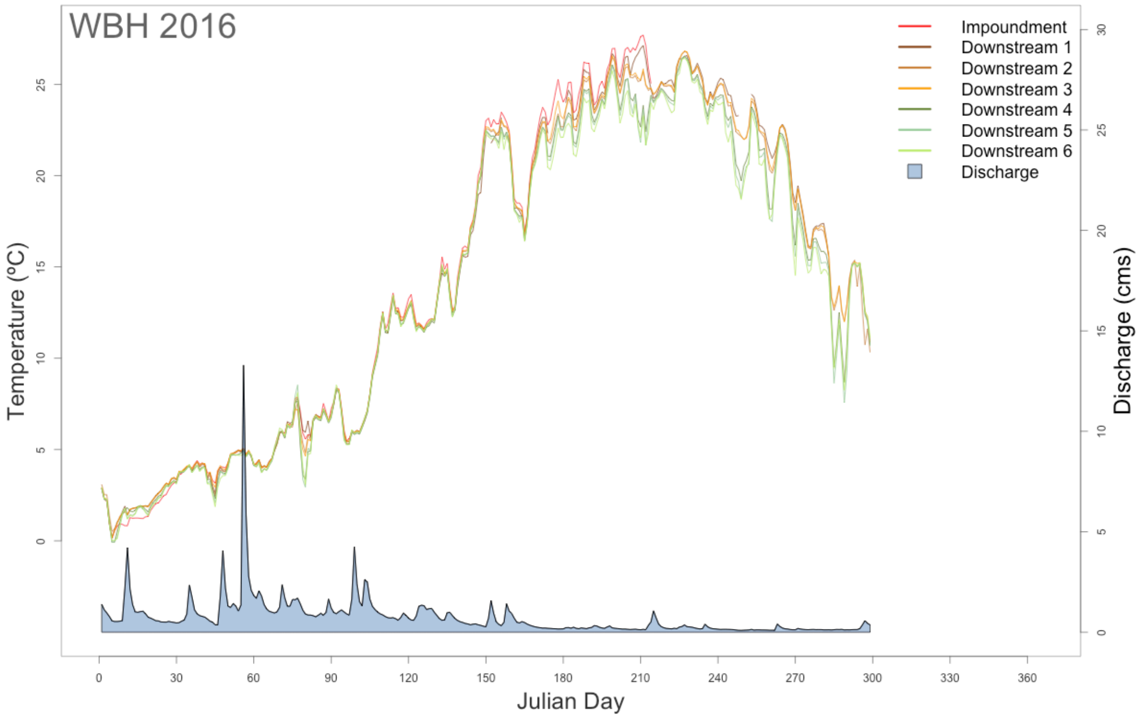
Mean daily temperature from the 7 loggers deployed upstream, within the impoundment, and downstream of Upper Roberts Meadow Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



Mean daily temperature from the 7 loggers deployed downstream of West Branch Housatonic / Pontoosuc Dam in 2014. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



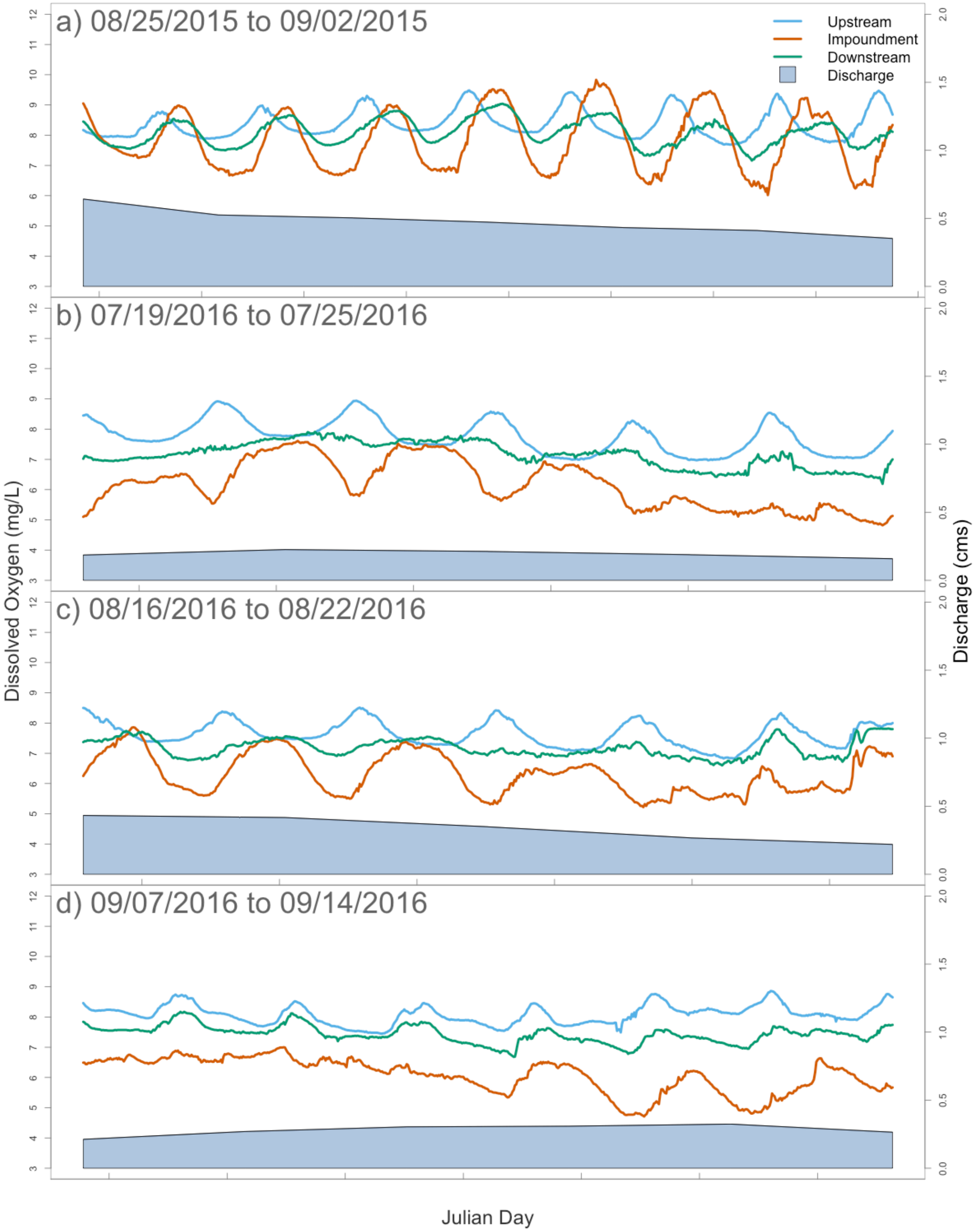
Mean daily temperature from the 7 loggers deployed within the impoundment and downstream of West Branch Housatonic / Pontoosuc Dam in 2015. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



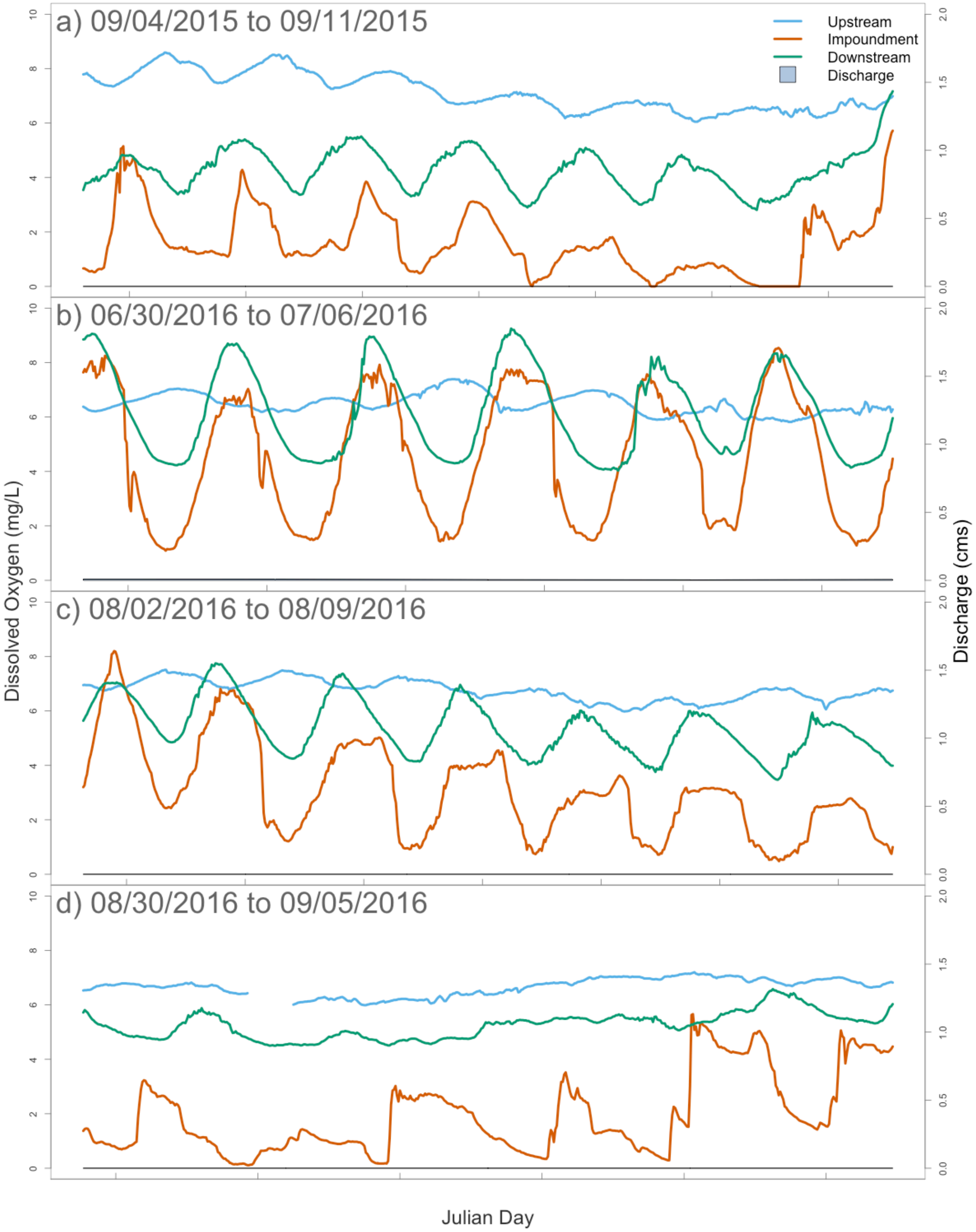
Mean daily temperature from the 7 loggers deployed within the impoundment and downstream of West Branch Housatonic / Pontoosuc Dam in 2016. Daily discharge (secondary y-axis) is plotted as the shaded polygon.

APPENDIX E

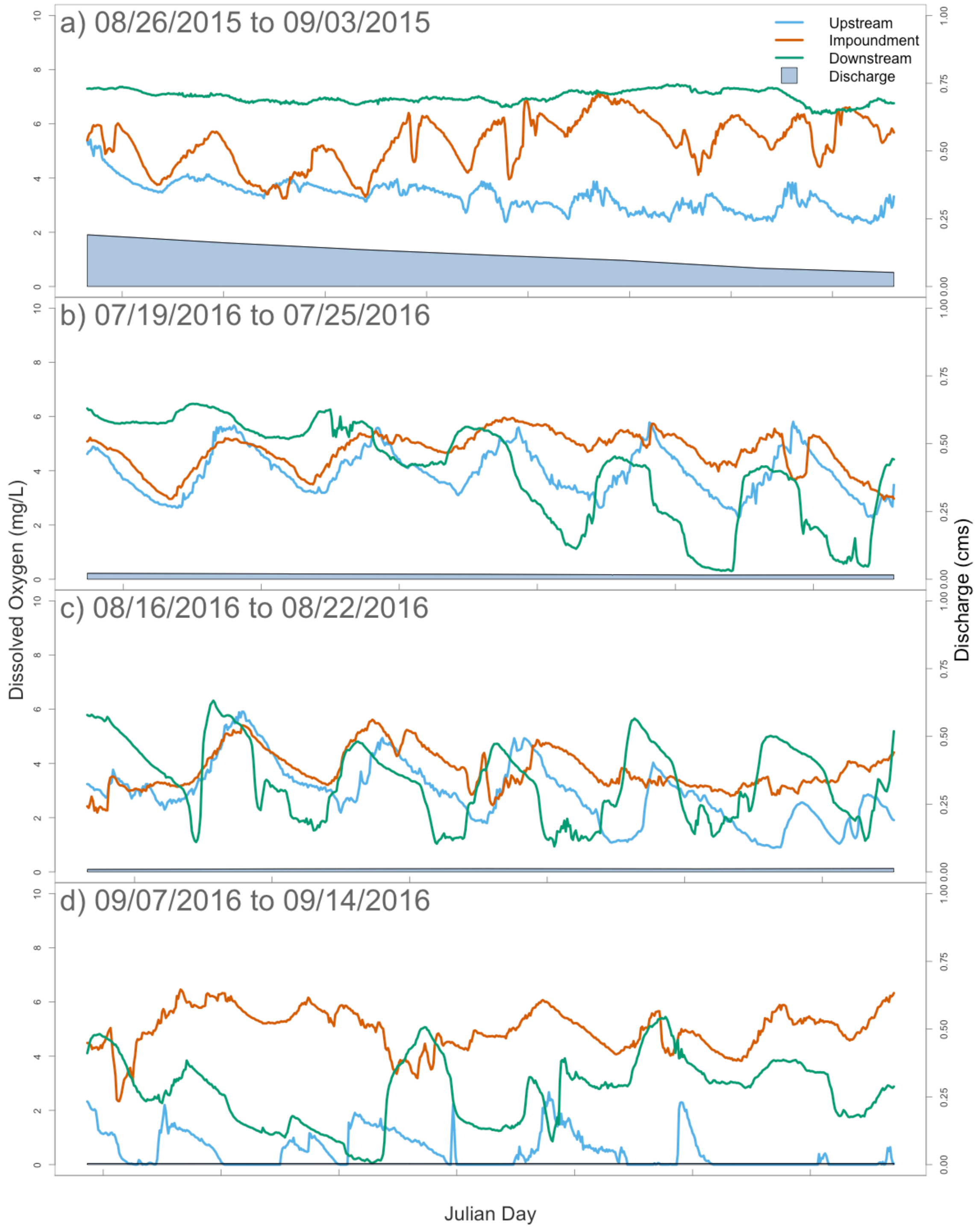
SITE-SPECIFIC CONTINUOUS PRE-DAM REMOVAL DISSOLVED OXYGEN PROFILES



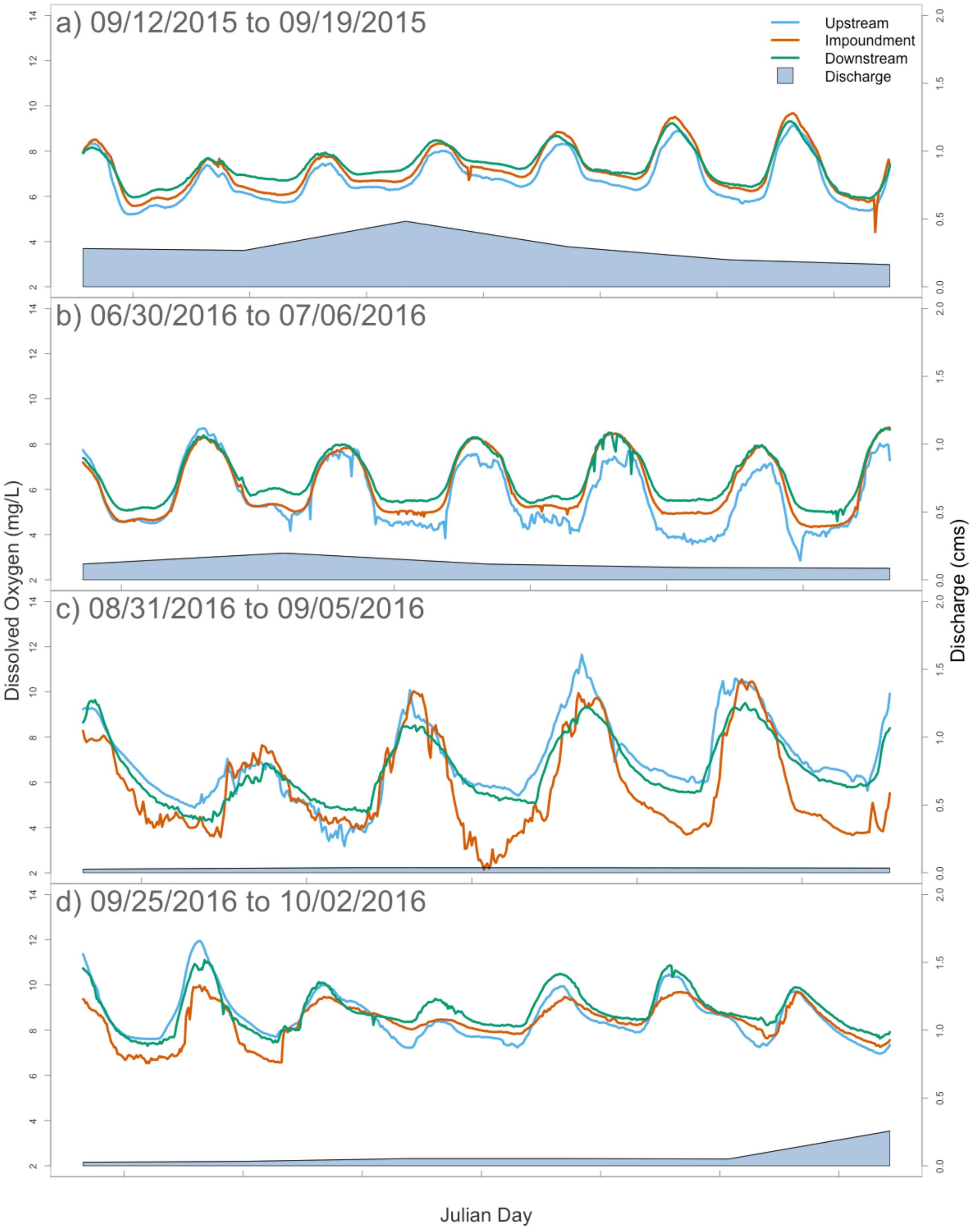
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Balmoral Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



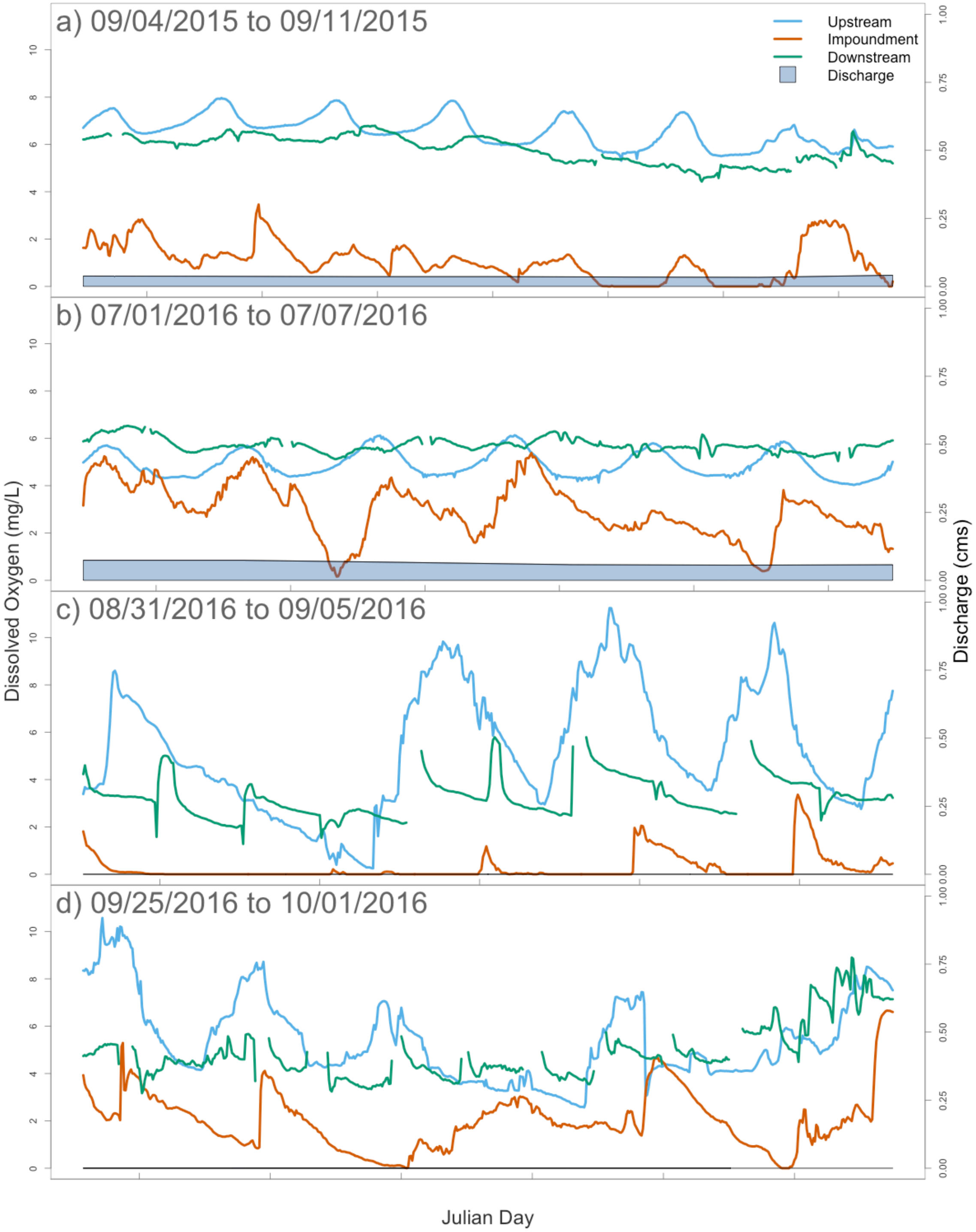
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Barstow's Pond Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



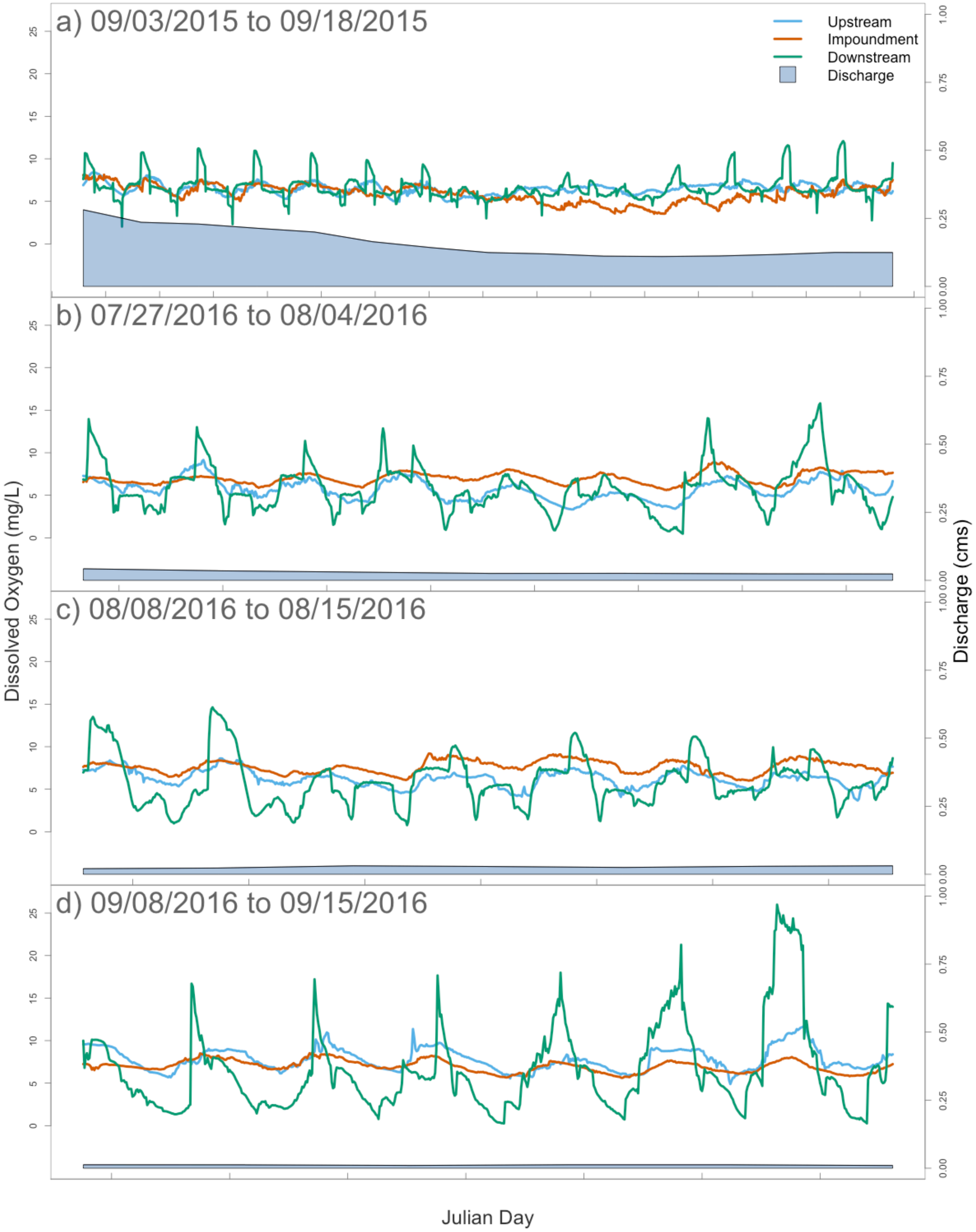
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Bostik / South Middleton Dam during the study’s four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



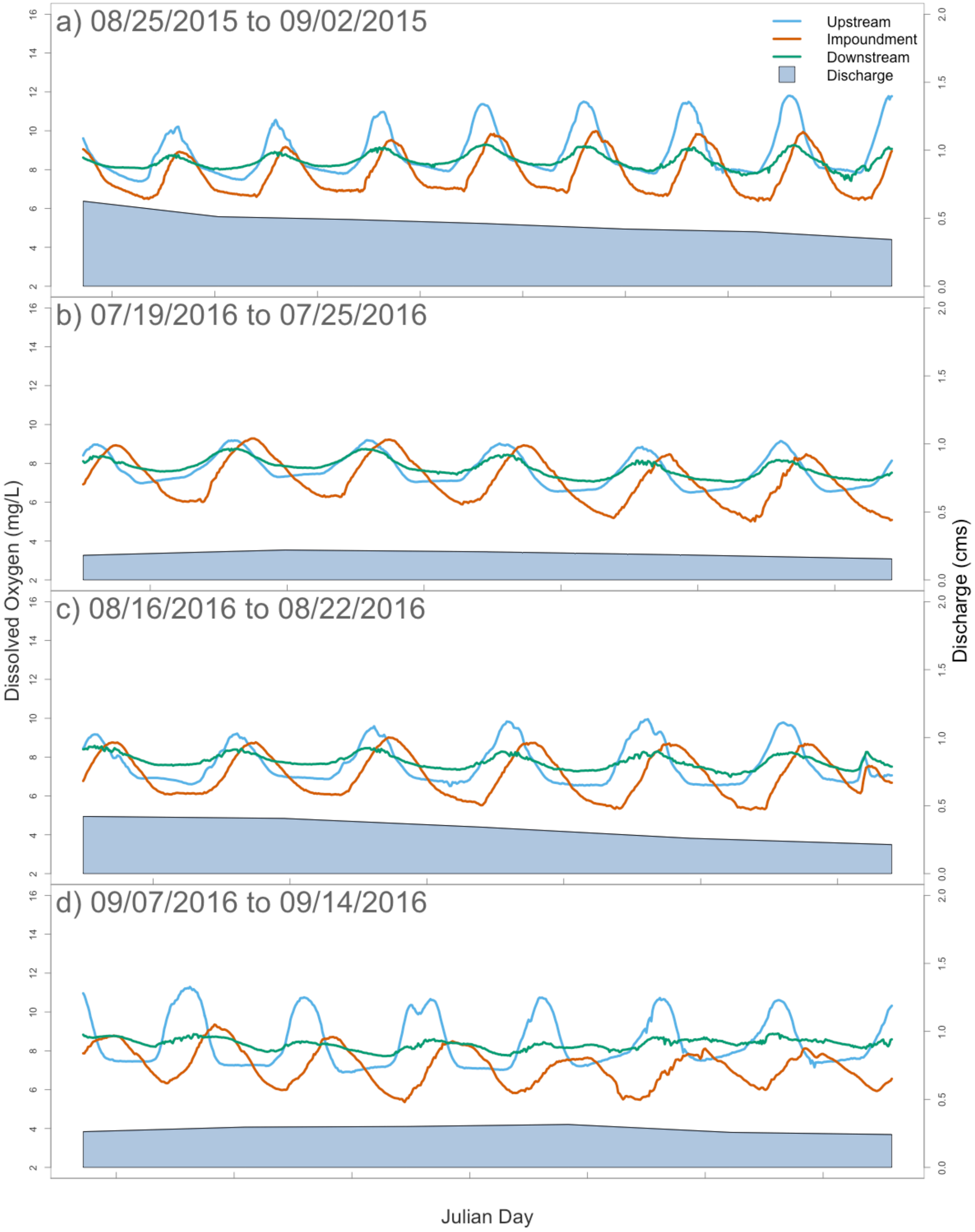
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Cotton Gin Mill Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



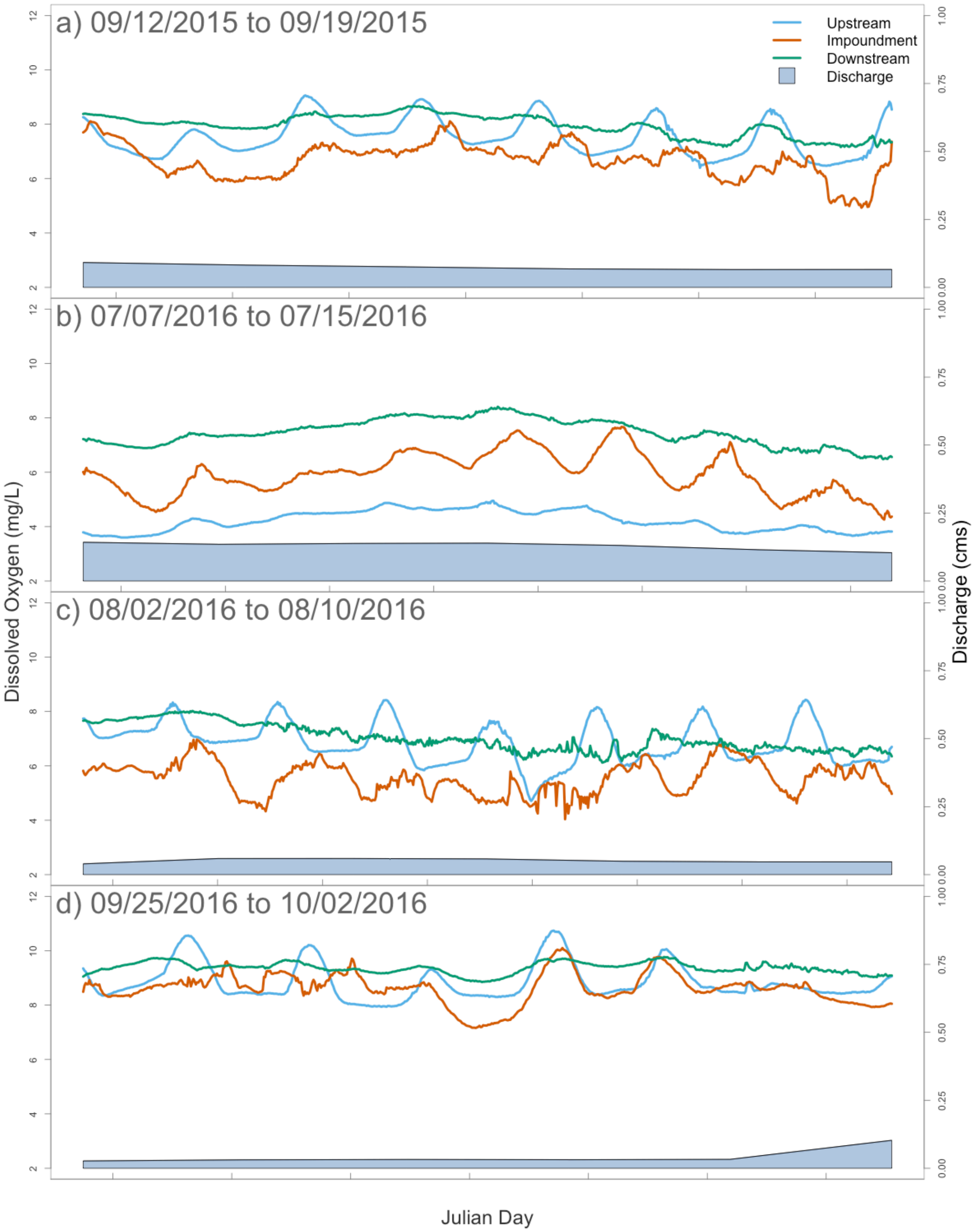
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Hunter's Pond Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



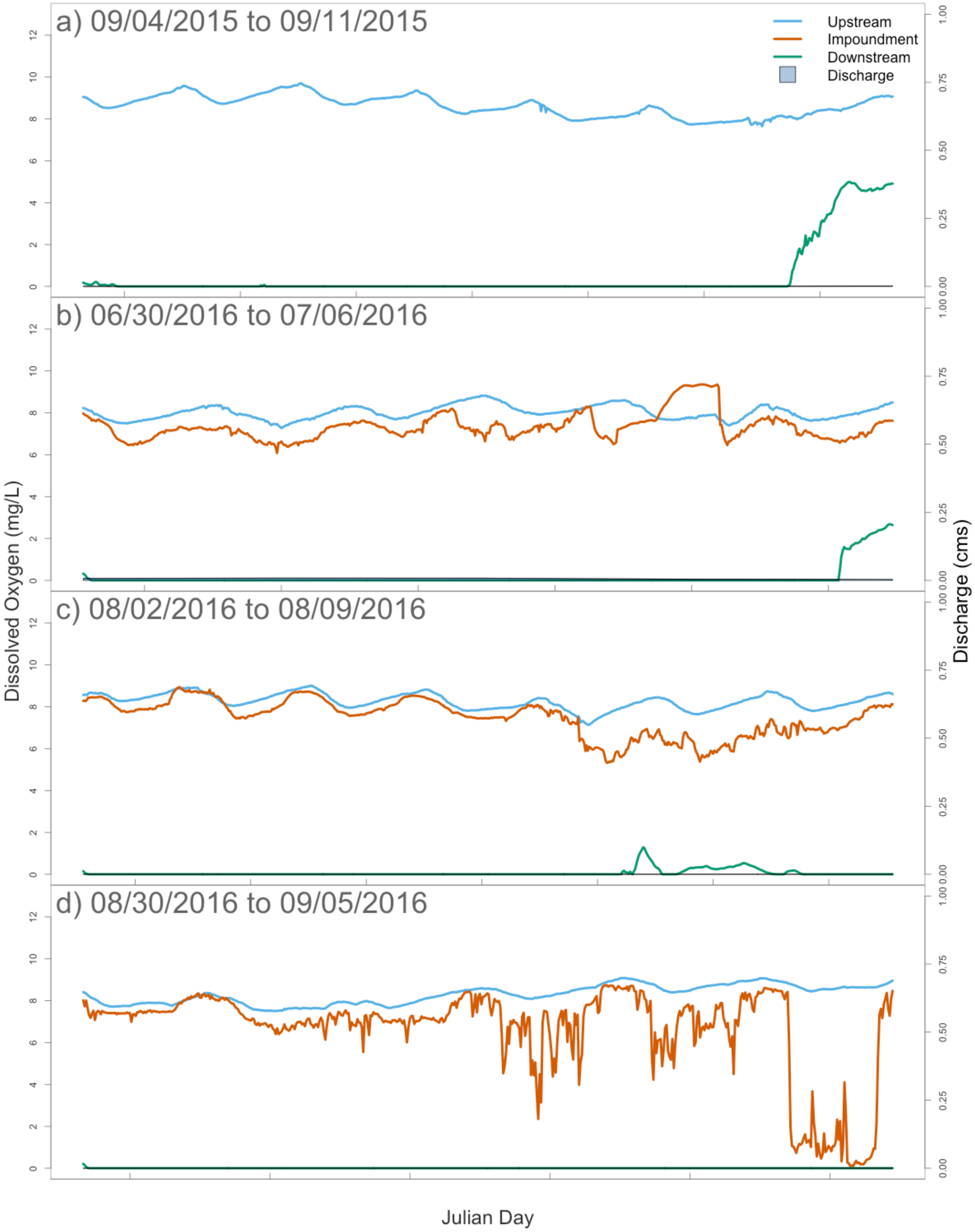
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Ipswich Mills Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



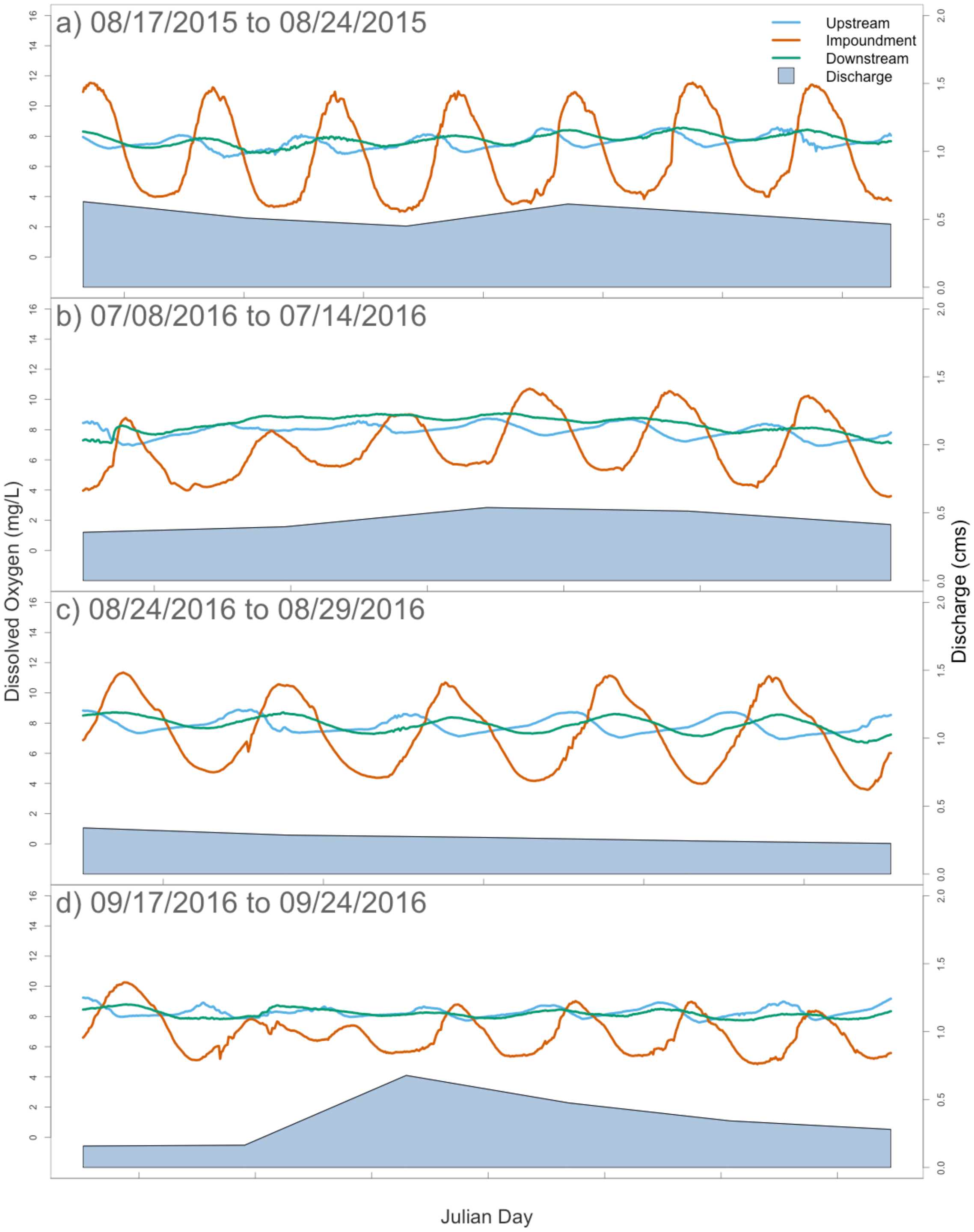
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Marland Place Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



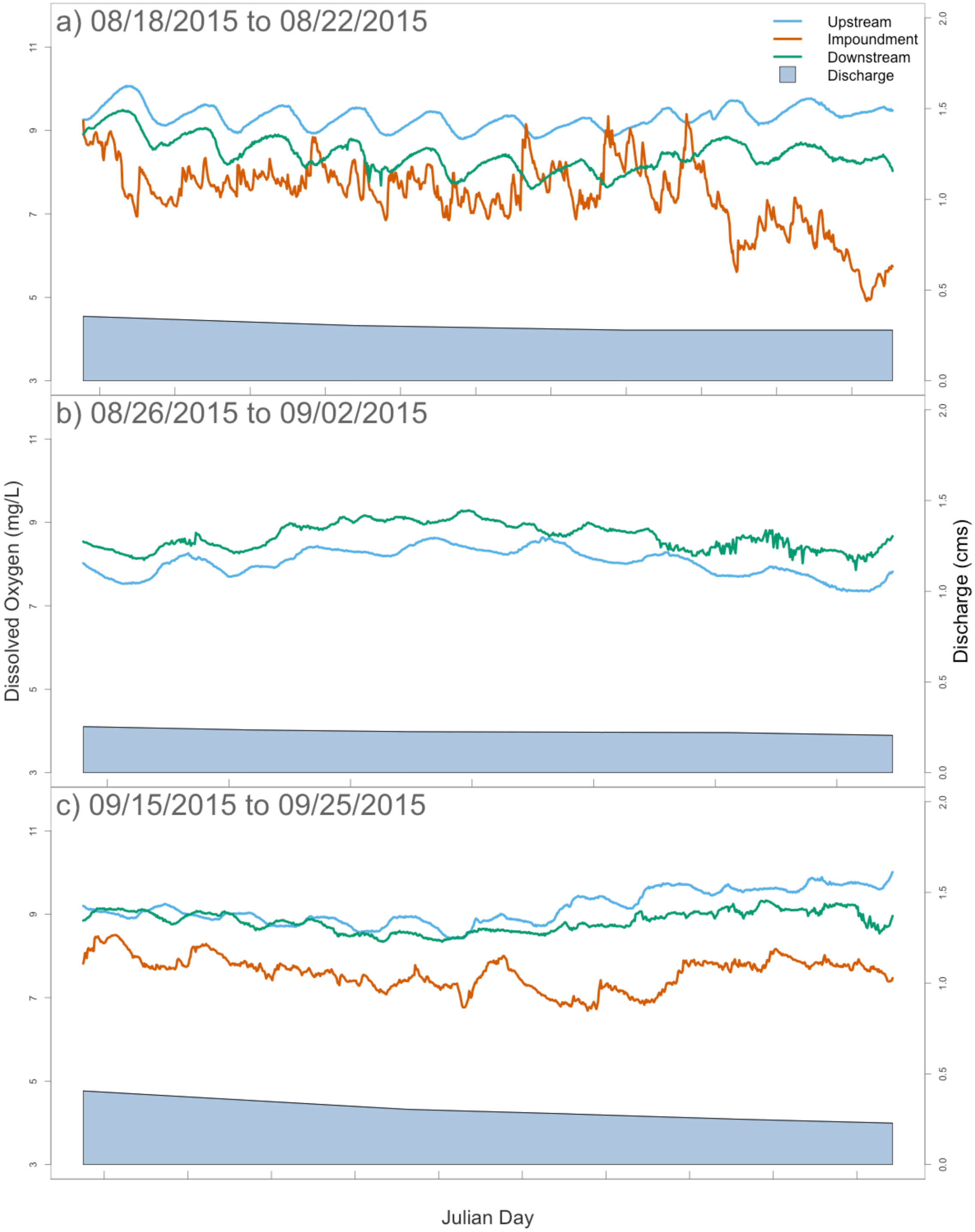
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Old Mill Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



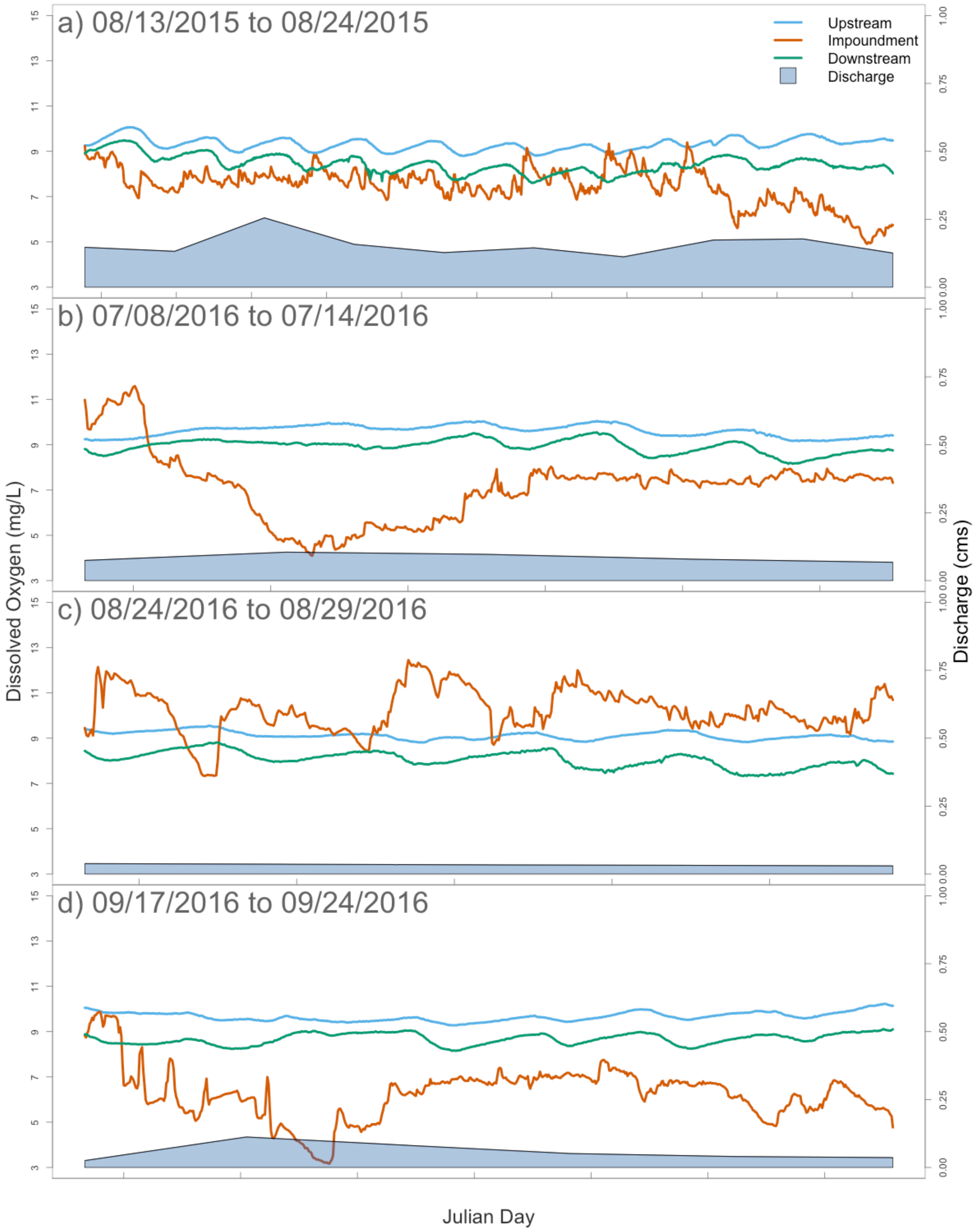
Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Rattlesnake Brook Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Tel-Electric Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Turner Dam during the study's three pre-dam removal deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.



Dissolved oxygen concentrations (recorded at 15-minute intervals) for each reach at Upper Roberts Meadow Dam during the study's four deployment periods. Tick marks on the x-axis represent each day of the deployment; note differing deployment lengths for the different deployments. Daily discharge (secondary y-axis) is plotted as the shaded polygon.

APPENDIX F

VERTICAL PROFILE MULTIPARAMETER READINGS TAKEN AT THE DEEPEST POINT OF EACH IMPOUNDMENT

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Balmoral impoundment. Depth (m) is distance below the surface.

Date Time	Depth (m)	DO (mg/L)	DO (%)	Temp (ºC)	pH (pH)	Cond. (us/cm)
9/2/15	0.0	8.07	95.6	23.70	7.47	647
@ 19:30	0.5	7.95	94.0	23.45	7.33	644
	1.0	6.66	77.7	22.82	7.23	642
7/25/16	0.0	4.57	54.7	24.73	6.93	761
@ 12:07	0.5	4.59	55.1	24.42	6.93	764
	1.0	4.31	51.3	24.21	6.90	753
	1.4	2.16	25.2	23.49	6.74	737
8/22/16	0.0	6.59	76.0	22.45	6.70	246
@ 11:12	0.5	6.97	80.0	22.41	6.63	247
	1.0	7.08	81.2	22.25	6.59	238
	1.3	6.70	76.7	22.22	6.53	237
9/14/16	0.0	5.47	61.2	20.72	6.99	690
@ 15:30	0.5	5.18	57.3	20.20	6.95	693
	1.0	4.45	49.0	20.00	6.88	693
	1.3	4.38	48.0	19.93	6.82	696

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of Barstow's Pond. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
9/11/15	0.0	6.04	68.0	20.78	6.58	318
@ 13:45	0.5	5.41	60.9	20.77	652.00	319
7/6/16	0.0	7.61	90.2	23.68	6.74	510
@ 13:30	0.5	4.80	54.6	21.71	6.60	510
8/9/16	0.0	1.60	18.3	22.44	6.44	478
@ 10:51	0.5	0.62	7.0	21.74	6.35	480
9/5/16	0.0	6.47	70.3	19.68	6.72	444
@ 11:35	0.5	4.86	52.4	19.38	6.58	445

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Bostik / South Middleton impoundment. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
8/24/15	0.0	5.73	68.3	24.02	6.93	684
@ 14:45	0.5	5.11	60.0	23.27	6.85	684
	1.0	3.95	46.0	22.70	6.74	684
	1.5	2.51	28.5	22.29	6.55	673
	2.0	0.70	8.0	21.65	6.35	662
7/25/16	0.0	3.19	39.1	26.40	6.77	736
@ 14:18	0.5	3.08	37.7	25.54	6.76	736
	1.0	1.42	17.3	25.00	6.59	734
	1.5	0.30	3.4	23.86	6.53	720
8/22/16	0.0	4.35	53.2	25.25	6.72	694
@ 12:48	0.5	3.95	48.5	25.03	6.69	694
	1.0	3.98	49.7	24.92	6.67	694
	1.5	3.67	44.7	24.87	6.64	695
9/14/16	0.0	5.26	61.8	23.29	6.97	716
@ 17:11	0.5	4.75	55.6	23.16	6.93	714
	1.0	4.81	56.1	23.02	6.87	715
	1.5	1.70	19.2	21.16	6.51	710

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from two vertical profiles performed at the deepest point of Hunter's Pond. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
9/11/15	0.0	1.10	12.3	20.80	6.07	216
@ 11:00	0.4	0.59	6.6	20.64	5.98	216
7/7/16	0.0	2.22	25.9	22.94	5.71	207
@ 11:48	0.4	0.90	10.4	22.34	5.76	210

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Ipswich Mills impoundment. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
9/18/15	0.0	5.78	68.9	24.39	7.20	498
@ 15:10	0.5	5.45	64.9	24.32	7.18	499
	1.0	6.11	71.6	23.49	7.07	502
	1.5	4.00	44.9	21.38	6.74	573
8/4/16	0.0	7.32	90.1	26.83	7.34	528
@ 13:19	0.5	7.22	88.9	26.61	7.32	530
	1.0	7.73	94.3	25.51	7.30	541
	1.4	6.33	74.1	24.34	7.09	544
8/15/16	0.0	6.30	79.2	27.47	7.39	513
@ 13:27	0.5	6.41	80.4	27.16	7.37	511
	1.0	4.09	49.5	25.55	7.05	527
	1.4	1.84	21.9	24.82	6.78	549
9/15/16	0.0	6.64	75.7	22.18	7.55	595
@ 15:52	0.5	6.90	78.2	22.17	7.48	596
	1.0	6.52	73.1	21.51	7.37	598

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Marland Place impoundment. Depth (m) is distance below the surface.

Date Time	Depth (m)	DO (mg/L)	DO (%)	Temp (ºC)	pH (pH)	Cond. (us/cm)
9/2/15	0.0	8.30	98.9	24.10	7.69	639
@ 15:00	0.5	8.30	98.7	24.02	7.65	639
	1.0	8.14	96.4	23.76	7.55	640
	1.5	7.54	87.4	22.62	7.14	641
	2.0	6.76	77.6	22.13	6.97	641
	2.5	5.95	68.2	22.04	6.92	642
7/25/16	0.0	4.35	51.7	24.38	6.99	765
@ 10:27	0.5	4.46	53.0	23.63	6.98	762
	1.0	4.77	55.0	23.52	6.95	762
	1.5	4.55	53.7	23.43	6.90	762
	2.0	4.47	53.2	23.40	6.88	762
8/22/16	0.0	6.43	74.6	22.57	6.69	449
@ 9:47	0.5	6.55	76.2	22.55	6.67	442
	1.0	6.51	75.1	22.58	6.64	448
	1.5	6.50	75.3	22.57	6.64	455
	2.0	6.31	73.1	22.57	6.62	459
9/14/16	0.0	6.66	76.2	22.30	6.89	686
@ 14:12	0.5	6.22	69.2	20.90	6.87	685
	1.0	6.03	66.3	19.97	6.83	686
	1.5	5.84	63.9	19.70	6.82	688
	2.0	4.71	51.3	19.34	6.75	689
	2.5	4.51	48.9	19.24	6.70	690

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Old Mill impoundment. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
9/19/15	0.0	6.58	79.6	23.07	7.46	1038
@ 15:30	0.5	5.43	62.5	22.15	7.32	1048
	1.0	5.14	58.4	21.47	7.23	1047
7/15/16	0.0	3.83	46.3	24.61	6.92	756
@ 9:45	0.5	3.53	42.4	24.30	6.92	762
	1.0	3.53	42.3	24.22	6.89	762
8/10/16	0.0	4.30	51.0	23.90	7.17	1038
@ 10:17	0.5	4.22	50.1	23.89	7.14	1038
10/2/16	0.0	7.61	75.0	14.38	7.25	893
@ 13:26	0.5	7.51	73.5	14.26	7.24	891

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Tel-Electric impoundment. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
8/24/15	0.0	3.45	45.7	22.40	7.55	323
@ 10:30	0.5	3.80	45.0	22.24	7.54	322
	1.0	3.56	42.2	22.20	7.53	322
	1.5	3.33	39.5	22.20	7.52	322
7/14/16	0.0	3.13	38.3	23.50	7.44	300
@ 9:40	0.5	3.23	39.1	23.47	7.40	300
	1.0	3.18	38.7	23.42	7.39	301
	1.5	3.42	41.6	23.36	7.36	301
8/29/16	0.0	7.18	88.0	24.11	7.77	313
@ 13:53	0.5	6.96	85.2	24.12	7.76	312
	1.0	7.32	89.4	24.07	7.74	312
	1.5	7.32	88.7	23.63	7.82	310
9/24/16	0.0	5.92	63.8	17.67	7.63	273
@ 11:39	0.5	5.88	63.0	17.44	7.61	272
	1.0	5.95	63.8	17.42	7.56	272
	1.5	5.71	61.1	17.34	7.44	272

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from one vertical profile performed at the deepest point of the Turner impoundment. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
9/15/15	0.0	8.07	91.2	18.45	6.62	125
@ 14:15	0.5	8.17	91.8	17.90	6.63	124
	1.0	8.07	89.3	17.53	6.64	123

Multiparameter readings of dissolved oxygen (DO), temperature (Temp), pH (pH), and conductivity (Cond.) from four vertical profiles performed at the deepest point of the Upper Roberts Meadow reservoir. Depth (m) is distance below the surface.

Date Time	**Depth (m)**	**DO (mg/L)**	**DO (%)**	**Temp (ºC)**	**pH (pH)**	**Cond. (us/cm)**
8/24/15	0.0	6.18	71.8	22.06	6.90	90
@ 13:45	0.5	6.14	67.3	19.42	6.78	89
	1.0	5.51	59.3	18.46	6.71	90
	1.5	5.45	58.5	18.32	6.72	91
	2.0	4.67	50.0	18.20	6.59	91
	2.5	1.93	20.6	17.87	6.37	91
	3.0	0.00	0.0	16.41	6.42	111
	3.5	0.00	0.0	15.60	6.51	125
7/14/16	0.0	6.05	70.0	21.83	6.58	94
@ 13:07	0.5	6.05	69.2	21.18	6.49	95
	1.0	6.88	74.4	18.58	6.45	94
	1.5	5.17	54.5	17.41	6.30	96
	2.0	2.34	24.4	16.61	6.08	97
	2.5	0.00	0.0	15.65	5.95	100
	3.0	0.00	0.0	13.93	5.99	114
	3.5	0.00	0.0	12.08	6.08	127
	4.0	0.00	0.0	10.68	6.20	158
8/29/16	0.0	8.23	98.4	23.78	7.35	106
@ 19:11	0.5	9.23	104.9	21.10	7.35	107
	1.0	8.91	98.1	19.70	6.98	102
	1.5	4.74	51.3	18.89	6.44	102
	2.0	0.00	0.0	18.00	5.95	102
	2.5	0.00	0.0	17.19	5.77	109
	3.0	0.00	0.0	16.35	5.75	130
	3.5	0.00	0.0	14.74	5.84	161
	4.0	0.00	0.0	13.70	6.00	200
9/24/16	0.0	4.59	48.0	17.08	6.61	110
@ 14:19	0.5	4.43	45.7	16.41	6.59	110
	1.0	4.46	45.9	16.27	6.52	110
	1.5	2.90	29.8	16.19	6.17	111
	2.0	0.00	0.0	15.88	5.94	112
	2.5	0.00	0.0	15.52	5.83	116
	3.0	0.00	0.0	15.26	5.80	123
	3.5	0.00	0.0	14.94	5.81	143
	4.0	0.00	0.0	14.38	5.92	198

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