TRADEOFFS OF WATER SECURITY & ADEQUATE ECOLOGICAL PROTECTION DURING DROUGHT, COASTAL CALIFORNIA

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In partial fulfillment of
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In

Geography: Resource Management and Environmental Planning

by

Kelly Ann Krotcov

San Francisco, California

May 2019

CERTIFICATION OF APPROVAL

I certify that I have read Tradeoffs of Water Security & Adequate Ecological Protection During Drought, Coastal California by Kelly Ann Krotcov, and that in my opinion this work meets the criteria for approving a thesis submitted in partial fulfillment of the requirement for the degree (Master of Arts in Geography: Resource Management and Environmental Planning at San Francisco State University.

Leora Nanus, Ph.D.
Assistant Professor

Jerry D. Davis Ph.D.
Professor

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Kelly Ann Krotcov
San Francisco, California
2019

Instream flow is an important limiting factor influencing aquatic health. Under a decentralized watershed management regime, private reservoir storage is being promoted in a coastal California watershed to maintain ecological protections and water security during low-flow conditions. This study establishes a method for determining the annual water diverted for current private reservoir storage by using GIS methods to determine the maximum change in storage that is then compared to publicly available user reported water use data. In addition, this study investigates streamflow during low-flow conditions and finds that drought conditions tended to prolong the end of stream intermittency and that private reservoir recharge has the potential to extend intermittency. Finally, instream water temperature data was evaluated; temperatures were often not conducive to salmonid growth and often not even to their survival. Water temperature data also showed trends of lower temperatures during dry-type years and higher temperatures during wet-type years

I certify that the Abstract is a correct representation of the content of this thesis.

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Chair, Thesis Committee Date

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

GLOSSARY OF TERMS

Diversion: Refers to the take of water from either surface, instream, or ground water sources for human use.

Stream intermittency: A period of zero discharge that can persist during the dry season (Deitch, 2006). In coastal California it can occur as early as late spring following the end of the rainy season and ends with the onset of the rainy season during the following rainy year. While a period of natural stress for aquatic life there can be pools that persist throughout this period which are sometimes maintained by trickles of localized base flow. The pools allow aquatic life to persist until the rainy season resumes and restores instream flow.

Impairment: Herein refers to the impaired discharge that occurs from filling private reservoirs with surface flow at the onset of the rainy season and is a metric for investigating changes in timing and the potential extension of the end of intermittency.

Water year: A term used by the USGS to define a 12-month period in reference to surface-water supply which in any given year ranges from October 1st through the following September 30th. "The water year is designated by the calendar year in which it ends and which includes 9 of the 12 months." (USGS, 2016)

1.0 INTRODUCTION

1.1 DROUGHT

Coastal California is characterized as a Mediterranean climate, which experiences cool, wet winters and dry, warm to hot summers. The majority of the rainfall occurs between November through March (Grantham, Mezzatesta, Newburn, & Merenlender, 2014) with drought naturally occurring in the summer. Drought is considered a natural disturbance in fluvial systems (Lake, 2000) and even though biotic communities evolved with this regime, human alternation of the landscape and hydrologic processes affects the resiliency of aquatic ecosystem health in fluvial systems (Magoulick & Kobza, 2003; Poff et al., 1997). There is a dichotomy of varying abiotic and biotic pressures on aquatic populations (Gasith & Resh, 1999). Fluctuations in seasonal streamflow, in addition to episodic rainfall events, alter between winter flooding and dry season intermittency which can vary over annual and multiannual scales influencing the function of freshwater ecosystems and community structure (Gasith & Resh, 1999).

Streamflow trends and precipitation rates follow rainfall patterns with higher magnitude discharge occurring in the winter months and then receding throughout the spring with some streams reaching a period of intermittency. Intermittency during the dry season is a period of zero instream discharge (Deitch, 2006). While intermittency is a period of natural stress for aquatic life there can be pools that persist throughout this period which are sometimes maintained by trickles of localized base flow. The pools allow aquatic ecosystems to persist until the rainy season resumes and restores instream flow.

1.2 WATER SECURITY

In coastal California watersheds, where there is not access to large-scale water projects, human water needs are met through local sources (Deitch, Kondolf, & Merenlender, 2009b). These local sources can include direct diversions from streams, groundwater pumping, or private small-scale reservoirs which can be filled through winter runoff or through diversions via the former methods (Deitch et al., 2009b; Grantham et al., 2014; Grantham, Newburn, McCarthy, & Merenlender, 2012) (*Figure 1*).

Figure 1: Example of small-scale instream surface water diversion. (Shafer, 2012)

While the term water security can be defined and used in many ways, one common discourse is water availability for human use and sometimes more specifically it can focus on a reliable and sufficient water supply to meet agricultural needs (Cook & Bakker, 2012). In dry regions, private reservoir storage is often discussed as part of the solution (Cook & Bakker, 2012). For the purposes of this thesis water security will be used to describe water needs for human use.

In coastal California wine-grape growers use irrigation to protect their crops against extreme air temperature conditions. Irrigation is used for both frost and heat protection to mitigate damage to crops (Deitch, Kondolf, & Merenlender, 2009a). Both methods use spray irrigation to irrigate wine-grape fields in order to protect the crop from damage. Deitch et al. (2009a) found that when air temperatures approached 0°C or increased above 32°C, streamflow was quicker to recede and they were able to correlate the dewatering of the stream with frost protection management practices in the Maacama watershed. In addition to instream diversions, shallow groundwater pumping in the Russian River Watershed can accelerate intermittency by reducing subterranean subsurface flow (NMFS, 2007b) as it often occurs near streams (Grantham et al., 2012).

1.3 ANADROMOUS SALMONIDS & ECOSYSTEM HEALTH

Salmonid populations in Northern Coastal California have been declining in recent years. This decline has led to coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) receiving protection by the endangered species act (ESA) (NMFS, 2007a; NOAA, 2012) along with other vulnerable aquatic species.

The Central California Coast (CCC) Evolutionarily Significant Unit (ESU) coho are on both the State and Federal Endangered Species Lists (CDFW, 2015). According to the National Oceanic and Atmospheric Administration (NOAA) Fisheries Service, the population of CCC coho salmon ESU has been declining since being listed federally on the ESA as of 1996 and the species is on the brink of extinction (*Figure 2*) (NOAA, 2012). Coho are not alone. The CCC steelhead ESU are listed as federally threatened along with other aquatic species including California freshwater shrimp, California red-legged frog, and the foothill yellow-legged frog that are federally threatened and endangered (SRCD, 2015).

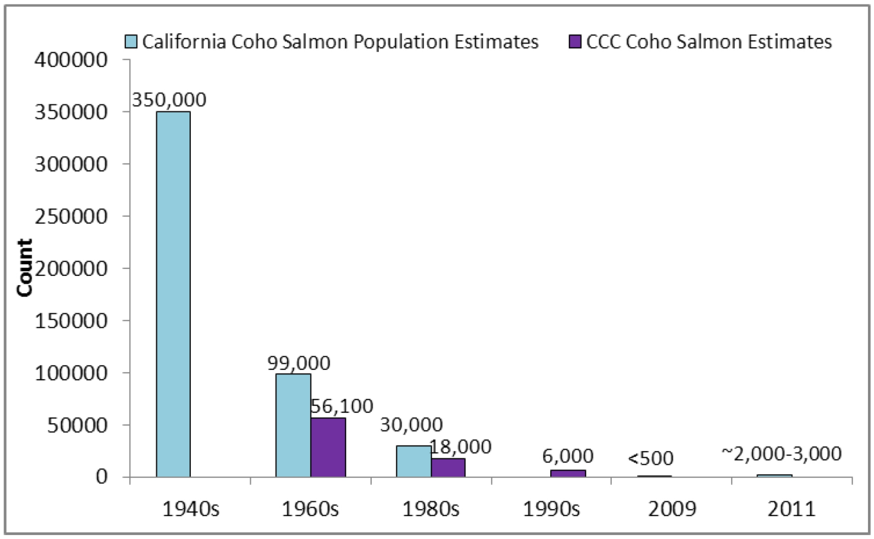


Figure 2: Historical estimate of spawning coho salmon (NOAA, 2012, p. viii)

Salmonids have specific habitat requirements throughout their lifecycle and habitat suitability is critical at each life stage (NOAA, 2012). Most salmonids are anadromous, hatching and rearing in freshwater then migrating to the ocean for their adult life, eventually returning to freshwater to spawn. While habitat requirements vary by species and life stage, they need adequate substrate along with appropriate water quality, quantity, temperature, and velocity (NMFS, 2008). Salmonids also require riparian vegetation along with adequate shelter, food and conditions that allow migration patterns (NMFS, 2008). Juvenile coho rear for just over a year before heading out to the ocean (NOAA, 2012) while juvenile steelhead can remain in the stream to rear for two or more years (NMFS, 2008) making it important to maintain adequate habitat suitability year-round. The peak of coho migration occurs in December and January (NMFS, 2008). Spawning occurs soon after the adults reach their spawning grounds (NMFS, 2008). The majority of steelhead migration occurs in January to March with spawning occurring in that same period (NMFS, 2008).

Maintaining instream flow is imperative for salmonids to persist into adulthood. While salmonids have adapted to the natural flow regime (Merenlender, Deitch, & Feirer,

2008) the dry season can put significant stress on the survival of rearing salmonids (Grantham et al., 2012; Magoulick & Kobza, 2003). The dry season has been shown to be equally as important to the survival of juvenile salmonids as is the wet season (Grantham et al., 2012). Deep pool habitat is essential to all life stages (NMFS, 2007b). During the dry season, hydrologic connectivity can be reduced or impaired, water temperatures can rise and pools can become isolated which can lead to overcrowding, predation, competition and stranding (Magoulick & Kobza, 2003). While rearing salmonids can survive in isolated pools during natural periods of intermittency (Grantham et al., 2012), it's a period of stress and reduced habitat quality (Poff et al., 1997). Deitch et al., (2009a) found that adequate streamflow was a limiting factor for survival during dry season conditions in the Maacama watershed.

1.4 CHALLENGES OF WATERSHED MANAGEMENT IN A DECENTRALIZED WATERSHED REGIME

> Arguably the greatest obstacle to environmental water allocations is reconciling conflicting human and ecosystem water needs.
>
> ~(Grantham et al., 2014, p. 315)

In contrast to large-scale water projects, decentralized small-scale water projects can present different benefits and challenges to ecological health. Small-scale water abstractions tend to use less water and are often unevenly distributed throughout the watershed potentially dispersing their impacts and in turn making it more attractive to environmental managers (Deitch et al., 2009b). While less understood or recognized compared with large-scale centralized water projects (Deitch et al., 2009a; Grantham et al., 2014), small-scale water diversions are not without impacts to the health of aquatic ecosystems. Effects of these systems can result in both spatial and temporal variability of streamflow (Deitch, 2006; Deitch et al., 2009b) and should be evaluated at fine temporal scales, like daily or sub-daily (Deitch et al., 2009b). Deitch et al. (2009a) found that there was localized and unsustained stream dewatering in response to small-scale water diversions for heat and frost protection during the dry season when streamflow was already naturally low in the Maacama watershed. They also found that spring and summer base flows were affected. Increasing the severity and duration of low flows during the dry season can lead to increased water temperatures, reduced dissolved oxygen, increased pollutants, and reduced food resources (Nilsson and Renofalt, Myrick and Cech, Harvey et al. and Hayes et al. as cited in Grantham et al., 2012, p. 595; Poff et al., 1997).

Conflicts of water security and adequate ecological protections are highest during periods of drought. At a time of naturally high stress for rearing salmonids (Poff et al., 1997), human water use demands rise with the onset of the dry season (Deitch et al., 2009b; Grantham et al., 2012). Small-scale water diversions have the potential to extend periods of dry season intermittency (Deitch, 2006; Deitch et al., 2009a). Grantham et al. (2012) found a significant correlation between increased vineyard land use and lowered rates of juvenile salmonid survival.

Conflicts are likely to increase with climate change impacts (Grantham, Merenlender, & Resh, 2010). Climate change is expected to exacerbate natural climate variability in coastal California (NCRP, 2014) likely intensifying negative impacts of water extractions and other anthropogenic stressors along with the duration and severity of natural disturbances (Palmer et al., 2009). Fluvial systems are complex in both their processes and response (Bendix & Hupp, 2000; Chin, Florsheim, Wohl, & Collins, 2013; Florsheim, Chin, Gaffney, & Slota, 2013; Gordon & Meentemeyer, 2006; Liebault & Piegay, 2002). Proactive environmental management strategies that anticipate changing climate and are able to adapt to those changes are important to the resiliency of a watershed (Palmer et al., 2009) demanding a multifaceted holistic management approach that considers ecosystem health alongside the socio-political factors that play into watershed degradation.

One watershed management strategy being promoted is the addition of private reservoirs to provide water security and promote streamflow during the dry season (*Figure 3*) (SRCD, 2015; Trout Unlimited & CEMAR, 2013). There are some complications to this movement, however. First, there has been a backlog of permitting of private reservoirs in part due to the ESA listing of coho salmon (Grantham et al., 2014). Second, well-intentioned policy can have unintended consequences. With the awareness that small-scale water projects have the ability to degrade streamflow, a narrative has arisen that the problem is less a lack of water but more an issue of management and timing of water diversions (RRCWRP, 2011); however, past studies have shown that water security can often bring increased growth and development as well (Piper, 2014; Walker & Williams, 1982; Worster, 1982).

Figure 3: Example of a small private reservoir. These types of reservoirs are being promoted in the watershed to protect summer instream flows (Merenlender et al., 2008)

1.5 RESEARCH QUESTIONS

Private reservoirs storage is being promoted in a decentralized watershed management regime as a way to maintain ecological protections while also maintaining water security during dry-type conditions. This study focuses on the tradeoffs of promoting private reservoir storage and maintaining ecological protections during extreme dry-type conditions by investigating the streamflow effects of private reservoir storage on streamflow with the following research questions:

1. Where are the existing reservoirs in the watershed and what is the estimated maximum capacity of those reservoirs? How does publicly reported water storage data compare to existing reservoirs in the watershed?
2. Are there spatial or temporal patterns of intermittency?
    a. Does intermittency begin earlier or end later following dry-type years?
    b. Is intermittency duration longer during dry-type years?
    c. Do years with likely frost or heat protection have any apparent effects on early season intermittency?
    d. Does the proportion of storage to watershed area affect the duration of intermittency?
    e. Is the duration of intermittency affected by the location within the drainage network?
3. What are the potential effects of private reservoirs early in the wet season?
    a. Does the location within the drainage network affect the end of impairment?
    b. How does precipitation timing and intensity affect impairment?
4. What is the likelihood of coho and steelhead survival based solely on instream water temperatures? Are there patterns of instream temperature based on dry-type and wet-type conditions?

2.0 METHODS

2.1 STUDY AREA

Once a “world-renowned steelhead and salmon fishery” the Russian River watershed has been reduced to just a few fish per year (Olin & Eckman, 2013, p. 1). The Russian River watershed runs south from Mendocino County and through Sonoma County until it turns west and terminates at the Pacific Ocean (*Figure 4*).

The Maacama watershed is a subwatershed of the Russian River watershed located in Sonoma County, California near the city of Healdsburg (*Figure 4*). The Maacama Creek watershed is located on the eastern side and lower half of the Russian River Watershed and along the western slope of the Mayacamas mountain range. The entire Maacama watershed encompasses about 182 km^2. The headwaters begin at the western side of the watershed at over 1000 meters above sea level (asl) at the tops of the Mayacamas mountain range and terminate at the confluence of the Russian River at about 40 meters asl.

The Maacama Watershed includes two main sub-watersheds that encompass both the Maacama Creek and its tributaries (Maacama Creek subwatershed) and the Lower Franz Creek and its tributaries (Franz Creek subwatershed) (*Figure 5*). The Maacama sub-watershed includes McDonnell Creek, Briggs Creek, Kellogg Creek, and Maacama Creek subwatersheds. The Franz Creek subwatershed includes Bidwell Creek, Upper Franz Creek and Lower Franz Creek. The confluence of these two watersheds occurs just upstream of the confluence with the Maacama Creek and the Russian River.

Agriculture is the dominant land use in the watershed, which includes wine-grape growing and cattle grazing (Laurel Marcus and Associates, 2004) and covers about 21 square kilometers (RRISRP, 2016). While wine-grape agriculture only occupies 7% of the watershed it’s the dominant user of water (Grantham et al., 2014, p. 317). Agriculture is mainly concentrated on Knights and Franz Valley (*Figure 5*) while grazing occurs on both hillsides and a part of the valley (RRISRP, 2016). There is some rural residential housing but no towns exist within the watershed (Laurel Marcus and Associates, 2004, p. 24).

Small-scale private reservoirs are being promoted in the region as a way to relieve pressures on instream flow during the dry season; this study aims to evaluate streamflow effects of those private reservoirs during dry-type, average-type and wet-type years.

Figure 4: Russian River Watershed

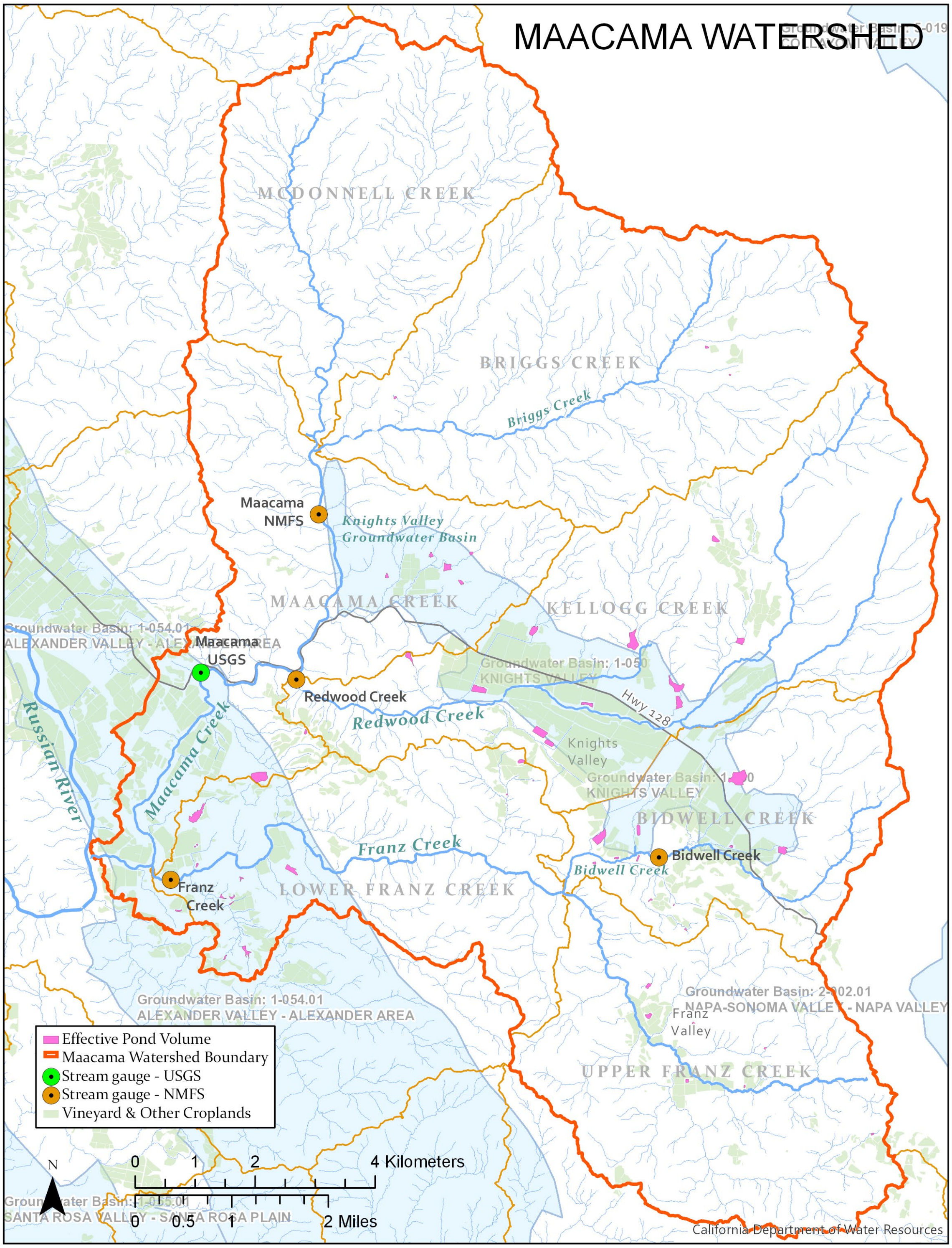


Figure 5: Maacama Watershed

2.2 DATA

2.2.1 GIS

The data included in this GIS analysis is listed below.

The majority of the data used for GIS analysis was acquired from the Sonoma County Vegetation Mapping and LiDAR Consortium (SCVMLC, 2018) which can be found at http://sonomavegmap.org/map_gallery/. Data used for analysis in this study included 1-foot interval contour (accessed 3/27/2017), croplands (accessed 4/15/2019), geology (accessed 4/15/2019), watershed boundaries (accessed 2/12/2018), high resolution orthophotos (accessed 9/30/2016), stream centerlines (accessed 2/12/2018), water and wetlands (accessed 2/12/2018) and a 1-meter Bare Earth Digital Elevation Model (DEM). The DEM was generated from LiDAR collected from 9/28/2013-11/26/2013 (WSI, 2016, p. 1). The LiDAR was "designed to yield high-resolution data of >8 pulses per square meter over terrestrial surfaces" (WSI, 2016, p. 12).

In addition to SCVMLC data, groundwater basin data was acquired from the California Department of Water Resources (accessed 4/16/2019), which can be found at https://gis.water.ca.gov/arcgis/rest/services/Geoscientific/i08_B118_CA_GroundwaterBasins/MapServer. The NMFS stream gauge locations were acquired from Chad Edwards of NMFS (C. Edwards, written communication, September 22, 2017). The USGS gauge location was acquired from the USGS (accessed 9/25/2017) and can be found at https://waterdata.usgs.gov/nwis/inventory/?site_no=11463900&agency_cd=USGS. Finally the parcel lot lines were acquired from the County of Sonoma (accessed 6/18/2018) and can be found at https://links.sonoma-county.org/nIhrCoQbqzY/.

2.2.2 HYDROLOGY AND CLIMATE

Discharge data was compiled from five stream gauges throughout the Maacama Watershed (see *Figure 5*). Discharge is expressed as a daily mean in this study. The

USGS 11463900 Maacama C NR Kellogg CA (a.k.a. Maacama USGS) gauge was publicly available through the United States Geological Survey (USGS) website (USGS, 2017). The Maacama Creek (a.k.a. Maacama NMFS), Redwood Creek and Bidwell Creek gauge data were acquired from NOAA's National Marine Fisheries Service (NMFS) (C. Edwards, written communication, September 22, 2017). The Franz Creek gauge data was acquired from a NMFS affiliate (A. Ticlavilca, written communication, December 14 & 21, 2018). All stream gauge data had an extrapolated daily mean discharge value (Q) calculated from a rating curve which was used for this research. All water and air temperature data was acquired with the Franz Creek gauge data.

Precipitation data was acquired from the Hawkeye weather station operated by Cal Fire (CDEC, 2017) (see *Figure 4*). The Cal Fire Hawkeye station is located in a watershed just under 7 km northwest of the Maacama watershed. It is the closest available weather station with data that is publicly available. The Cal Fire Hawkeye station is situated on the windward side of the Mayacamas mountain range similar to the Maacama watershed. By using this data it's assumed that the precipitation amounts for the Maacama watershed and each of its sub-watersheds are the same as at the Cal Fire Hawkeye station; however particularly in mountainous regions precipitation rates can vary over short distances.

Precipitation data from the Hawkeye Cal Fire station (7 km northwest of Maacama) was compared against the historical average derived from the Hawkeye RAWS weather station (15 km northwest of Maacama) (WRCC, 2019) to classify dry-type, average-type and wet-type years (see *Figure 4*). Water years with below average precipitation were classified as dry-type water years. Years with average precipitation was classified as an average-type year and years with above average precipitation was classified as a wet-type year. Precipitation data was also compared against the 'average maximum daily rainfall water-year' and the 'average annual rainfall intensity inches per days of rain' for historic data at the Healdsburg station (Laurel Marcus and Associates, 2004).

2.2.3 REPORTED WATER USE

Publicly available landowner reported water use data was obtained from the State of California Water Boards available through the Water Rights Information Management System (eWRIMS) online GIS mapping application (SWRCB, 2019a) and the eWRIMS Water Rights Records Search (SWRCB, 2015). The eWRIMS online GIS mapping viewer was used to search for an eWRIMS Application ID for each pond that had an EPV value. Orthophotography and parcel lot information were also compared in a GIS against the eWRIMS Application ID to help determine which ID went with which pond(s). After determining an ID, if applicable, the ID was used to look up the 'amount diverted or collected to storage' listed on past eWRIMS reports. Reports were listed annually by ID through the eWRIMS Water Rights Records Search engine.

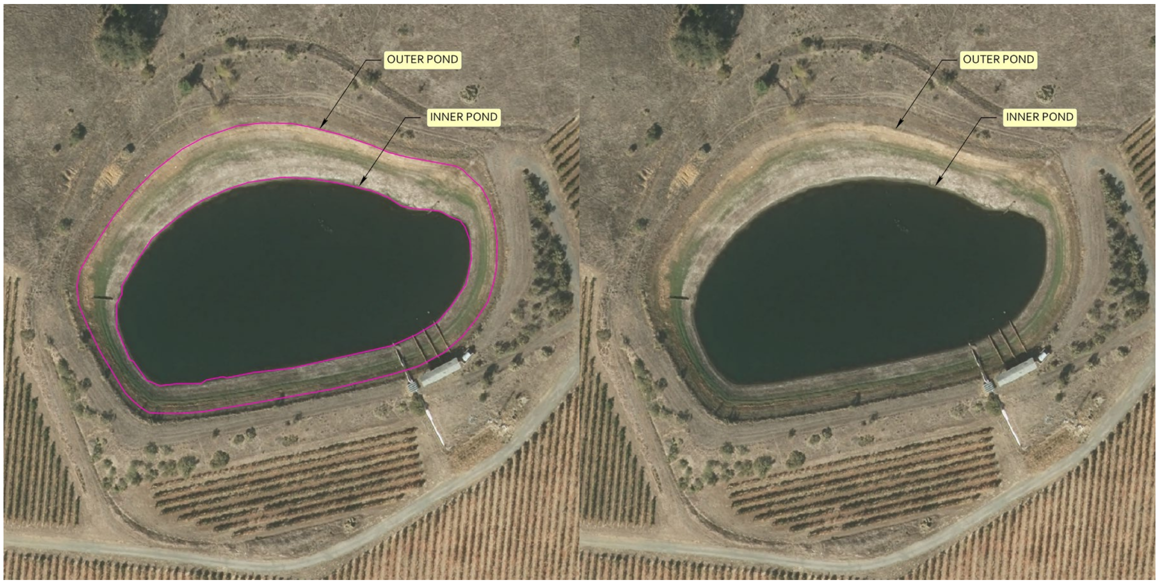
2.3 SPATIAL ANALYSIS

2.3.1 MANUAL DELINEATION OF PONDS

In order to digitize the existing private reservoirs (hereafter referred to as 'ponds') ArcGIS Pro 2.2.3 (ESRI, 2018) was used. The inputs used for identifying and digitally mapping the existing private reservoirs were 1) orthophotos 2) contours and 3) stream centerlines.

To locate the existing ponds, the basic areas of wetlands and ponds were evaluated. The data was then verified, altered, and digitized manually if needed using orthophotography, contour and stream centerlines data as a reference to visually interpret and delineate the edge of the ponds which resulted in an area and location for individual private reservoirs throughout the watershed. Because the orthophotography and LiDAR data were collected at the peak of the dry season during a drought year, both a minimum and maximum fill line was quantified for each pond. Ponds were excluded from the analysis if the inner and outer ponds were not quantifiable.

For this study, the inner pond represents the minimum amount of water in a pond. The inner pond was digitized at the water's edge at the water line as shown on the 2013 orthophotos below (*Figure 6*). The outer pond represents the maximum amount of water that can be stored in a pond. When digitizing pond locations visual indicators of human use or alteration of the water body were evaluated. Examples of the visual indicators included waterbodies that were lined with plastic or their shape appeared manmade (i.e. rectangular shape, straight side along water line or if it appeared to have a berm particularly located downstream along a creek). Other indicators included proximity to buildings and vineyards, pipes leading into reservoir, and barren sides with roads running alongside the pond. Indications of outer pond location included change in vegetation patterns that indicated a difference between an area that gets inundated by water periodically versus an area that appears terrestrial and is not influenced by periodic inundation of standing water. Typically, outer ponds were delineated where multiple indicators were found within less than a meter difference in elevation surrounding the current water line. The elevation of berms, pipes or buildings around a pond in which water storage was unlikely were also considered.



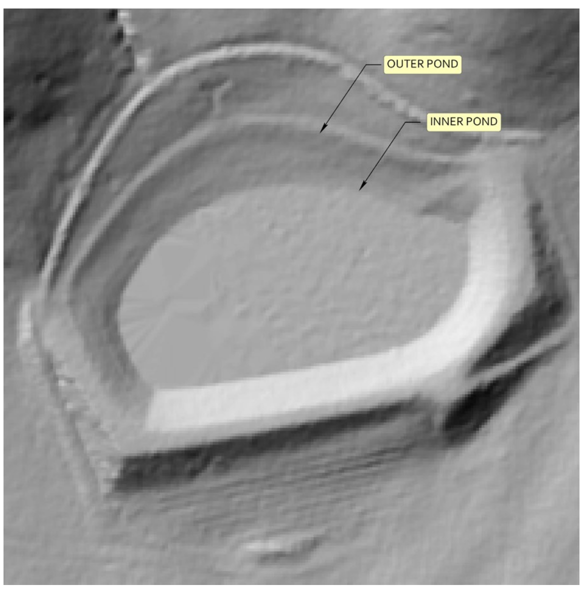


Figure 6: The photo on the *top left* shows the ortho aerial imagery with an example of the Inner Pond and Outer Pond polygons. In this photo you can also see pipes & a pump house indicating current or at least previous use. The photo on the *top right* shows the bare imagery. The photo on the *bottom left* shows LiDAR-derived shaded relief of the same pond as above.

2.3.2 CALCULATING EFFECTIVE POND VOLUME (EPV)

The existing reservoir locations and change in storage capacity were quantified using high resolution LiDAR data to locate existing private reservoirs and calculate the potential maximum amount of water being diverted to private reservoir storage annually. The LiDAR data was collected during the peak of the dry season during a drought which allowed an opportunity to quantify the change in reservoir capacity, hereafter referred to Effective Pond Volume (EPV). This approach builds on previous studies that used hypothetical ponds or estimated water use based on land use (Grantham, 2010; Merenlender et al., 2008).

To calculate the Effective Pond Volume (EPV), ModelBuilder (ESRI, 2018) was used within a GIS to calculate the change in volume between the outer and inner ponds of each respective pond (*Figure 7* and see *Appendix 1*). The inputs for the model were 1) a DEM and 2) a data layer of existing delineated ponds in the watershed with outer and inner pond polygons (see *section 2.3.1*).

```
# Clip DEM to Outer Pond
arcpy.management.SelectLayerByAttribute("Ponds", "NEW_SELECTION",
    "Inner_Outer_Pond = 'OUTER'")
arcpy.CopyFeatures_management("Ponds", "Ponds_selected")
arcpy.management.Clip("DEM", rectangle, "DEM_outer",)

# Fill Pond to Maximum
arcpy.PolygonToRaster_conversion("Ponds_selected", value_field,
    "Ponds_raster", "CELL_CENTER")
outZonalStats = ZonalStatistics("Ponds_raster", "Value",
    "DEM_outer", "MAXIMUM", "DATA")
outZonalStats.save("Ponds_maxFill")

# Calculate Change in Volume between Inner and Outer Ponds
arcpy.CutFill_3d("Ponds_maxFill", "DEM_outer", "EPV_CutFill", 1)

# Create EPV Feature Class & Add EPV Volume to Attributes
arcpy.RasterToPoint_conversion("EPV_CutFill", "EPV_RasterToPoint",
    "VOLUME")
arcpy.SpatialJoin_analysis("Ponds", "EPV_RasterToPoint",
    "EPV", "JOIN_ONE_TO_MANY", "KEEP_COMMON")
arcpy.AlterField_management("EPV", "grid_code", "Volume", "Volume",
    "LONG")
arcpy.AddField_management("EPV", "Volume", "FLOAT", None, 2, None,
    "Volume", "NULLABLE", "NON_REQUIRED", None)
```

Figure 7: Description of how to use a digital elevation model (“DEM”) and a feature class with inner and outer pond polygons (“Ponds”) to create an Effective Pond Volume feature class.

2.4 STATISTICAL ANALYSIS

2.4.1 EPV VS. REPORTED WATER USE

EPV is compared to user reported water use data during both a dry-type year and a average-type year. This builds upon work done by Deitch et al. (2009b) that uses user reported water use data to estimate water demand.

The volume of EPV was compared to the eWRIMS user reported volume of water use for both 2014 and 2016 using linear regression. RStudio v1.1.463 (RStudio, 2018) was used to create a linear model to test for spatial variability and determine the R-squared, p-values and residuals. The residuals were then compared using a GIS in order to look for geographic patterns or explanations for variability in the residuals. The linear models tested the EPV as the independent variable and the user reported water data as the dependent variable.

2.5 HYDROLOGIC ANALYSIS

2.5.1 WATERSHED DISCHARGE IMPAIRED BY PRIVATE RESERVOIRS

Temporal scale is important to consider when characterizing instream flow because both timing and duration are important to ecological processes (Deitch, 2006; Deitch et al., 2009b; Poff et al., 1997). Fine-scale analysis, like daily or sub-daily instream flow, is important to evaluating ecological protections for aquatic life (Deitch, 2006).

In order to obtain a fine-scale analysis of the potential effects that small-scale reservoirs have on instream flow, a modified water balance equation was used that was modified from Deitch et al. (2009b). Using the water balance equation, the impairment to discharge in response to filling private reservoirs upstream is calculated. Stream gauge data along with the EPV calculated above was used to determine the potential daily impairment the reservoirs have on streamflow as the rainy season resumes and the ponds are filled back to full capacity.

The inputs for this model were 1) mean daily streamflow at the stream gauge and 2) the total EPV above the gauge. The water balance equation modified from Deitch et al. (2009b) can be expressed as:

O (watershed discharge) = I (sum of upstream flow) – ΔS (sum of upstream diversions)

Where **I** is the measured flow at the gauge and Δ**S** is EPV. While Deitch et al. (2009b) used eWRIMS data to estimate the sum of upstream diversions (Δ**S)**: that approach may either under or overestimate water use (Deitch, van Docto, & Feirer, 2016; Grantham & Viers, 2014). This research builds upon the methods developed in Deitch et al. (2009b) by using the EPV value which locates private reservoirs based on high resolution LiDAR data in the watershed and establishes a consistent metric for determining the potential maximum amount of water being diverted for storage annually.

Typically small reservoirs in Coastal California operate by refilling at the beginning of the rainy season in turn collecting all discharge until the pond reaches maximum capacity (Deitch, Merenlender, & Feirer, 2013). The "fill-and-spill" technique, as it can be called, prevents downstream flow until the reservoir is full and only then is downstream discharge able to resume (Deitch et al., 2013).

To determine the reservoir-impaired flow at each gauge, the cumulative daily sum of upstream flow up to that point in the water year was compared against the total EPV above the gauge. Once the total upstream flow for the water year reached the total EPV above the gauge, the ponds were considered 'filled' and discharge resumed throughout the watershed.

2.5.2 STREAM INTERMITTENCY

The date of the start of intermittency was recorded on the first day of zero (0 m^3/s) discharge in the water year for each gauge. The first day of intermittency was defined as being after the last period of recorded discharge (>0 m^3/s) with at least 5 consecutive days of discharge. The end of intermittency was recorded as the last day of zero discharge (0 m^3/s) following the period of intermittency.

2.5.3 WATER TEMPERATURE & LIKELY FISH SURVIVAL

In addition to flow, water temperature can be an important limiting factor for salmonid survival during all life stages. Water temperature at the Franz Creek gauge was compared to coho and steelhead temperature growth and survival thresholds per the Russian River Watershed Biological Opinion (NMFS, 2008).

Preferred and lethal temperature limits vary by species and life stage. Juvenile coho salmon prefer water temperatures of 10-15°C (Bell and McMahon, as cited in NMFS, 2008, p. 53) which are good for "survival and growth" and at 18°C "growth slows considerably" (Stein et al. and Bell, as cited in NMFS, 2008). Rearing juvenile steelhead prefer water temperatures ranging from 7.2-14.4°C (NMFS, 2008). Even though under the right conditions steelhead can survive short periods in up to 27°C, water temperatures of 23.9°C is considered the general "upper lethal limit" to juvenile survival (NMFS, 2008).

3.0 RESULTS

3.1 SPATIAL ANALYSIS

The Maacama watershed has 54 ponds with a total EPV of 1,334,627 m^3 which includes 3 ponds below both the Franz Creek and Maacama USGS gauges (*Figure 5 and Table 2*). The area of the existing ponds ranges from 316 m^2 to 49,637 m^2 (*Table 1*). The average pond size is 8,771 m^2, with a median pond area of 2,875 m^2. The total pond area is 473,651 m^2.

	Pond Area (m^3)*
Range Min	316
Range Max	49,637
Median	2,875
Average	8,771
Total Pond Area	473,651

** Area of inner pond which is existing water line in 2013 orthophotography. n = 54*

Table 1: Pond area

The elevation of the stream gauges ranged from about 128 meters above sea level (asl) to the lowest in elevation at 46 meters asl (*Table 2*). The Maacama watershed comprises of two subwatersheds. The Maacama subwatershed has the Maacama NMFS, Redwood and Maacama USGS stream gauges. The Franz subwatershed has the Bidwell and Franz gauges. The gauge with the largest watershed area above it is Maacama USGS which is 57 meters asl, comprises 62% of the Maacama watershed, and has 48% of the total EPV in the watershed (*Figure 5* and *Table 2*).

	Total Watershed Area Above Gauge (km^2)	Elevation (meters asl)	Percent of Entire Watershed	Total EPV (m^3)	% of Total EPV	# of Ponds with EPV Value	EPV / Watershed Area (m^3/1 km^2)
Total above all gauges	174.4	-	96%	1,218,310	91%	51	-
Total below all gauges	7.1	-	4%	116,317	9%	3	-
Watershed Total	181.5	-	100%	1,334,627	100%	54	-
Above Maacama USGS Gauge*							
Maacama NMFS	61.2	85	34%	10,733	1%	3	175
Redwood Creek	33.2	68	18%	551,979	41%	11	16,647
Maacama USGS	112.8	57	62%	647,215	48%	19	5,736
Above Franz Creek Gauge**							
Bidwell Creek	18.5	128	10%	194,553	15%	3	10,531
Franz Creek	61.5	46	34%	571,095	43%	32	9,281
** Includes McDonnel Creek , Briggs Creek, Kellogg Creek & a portion of the Maacama Creek*							
***Includes Bidwell Creek, Upper Franz Creek, & a portion of the Lower Franz Creek subwatersheds*							

Table 2: Summary of watershed area & EPV relative to stream gauges

3.2 STATISTICAL ANALYSIS OF EPV TO REPORTED LANDUSE

Of the 54 ponds with EPV there were 24 ponds that had apparent corresponding reported water use data for both the 2014 and 2016 water years. When comparing the EPV to the user reported data for the corresponding ponds, the user reported water storage was 31% higher than the EPV during the 2014 water year and 19% higher than the EPV during the 2016 water year (*Table 3*). The user reported data was higher (10%) in the 2014 year than during the 2016 year. When comparing the EPV to the 'Face Amount,' which is the maximum allocated amount of water allowed for that individual permit by the state, the face amount was much higher than the EPV (60%) for the corresponding ponds.

	2014 (Dry-type year)		2016 (Average-type year)		Permitted	
EPV (m^3)	User Reported* (m^3)	Reported / EPV	User Reported* (m^3)	Reported / EPV	Face Amount (m^3)	Permitted / EPV
1,034,418.4	1,358,228	131%	1,232,964	119%	1,651,139	160%
**Amount diverted or collected to storage. n=24*						

Table 3: Summary of eWRIMS reported water use compared to the EPV for the corresponding ponds. Water use value shown is the 'amount diverted or collected to storage"

EPV was significantly related to the user reported data during both the 2014 and 2016 water years. The 2014 year had an R-squared = 0.54 (p-value< 0.01) and the 2016 year had an R-squared = 0.39 (p-value< 0.01) (*Figure 8*).

Residuals were estimated in RStudio (RStudio, 2018). Most of the ponds with the highest residuals also tended to have the largest area for both years (*Figure 9*). There were four ponds with residual values of ≥ 60,000 or ≤ -60,000 which occurred both years for three of the four ponds. There were more ponds in 2016 that were either ≥ 30,000 or ≤ -30,000 than in 2014. There were no obvious spatial patterns related to high residuals which included the proximity to the stream, location within the drainage network, or proximity to croplands. The ponds with higher residuals tended to be larger ponds.

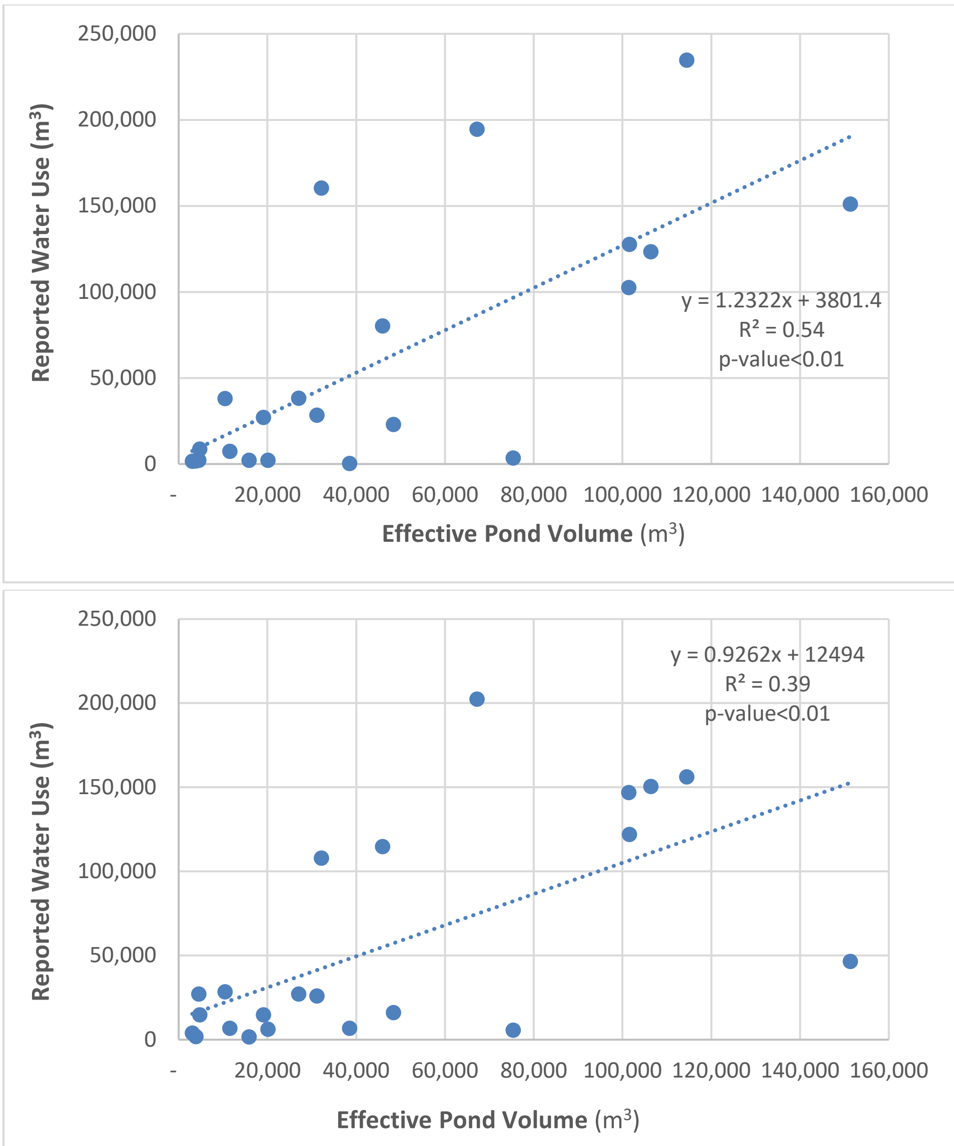


Figure 8: Plotted linear regression for EPV (m^3) vs. Reported Water Use (m^3) for the 2014 (top) and 2016 (bottom) water years

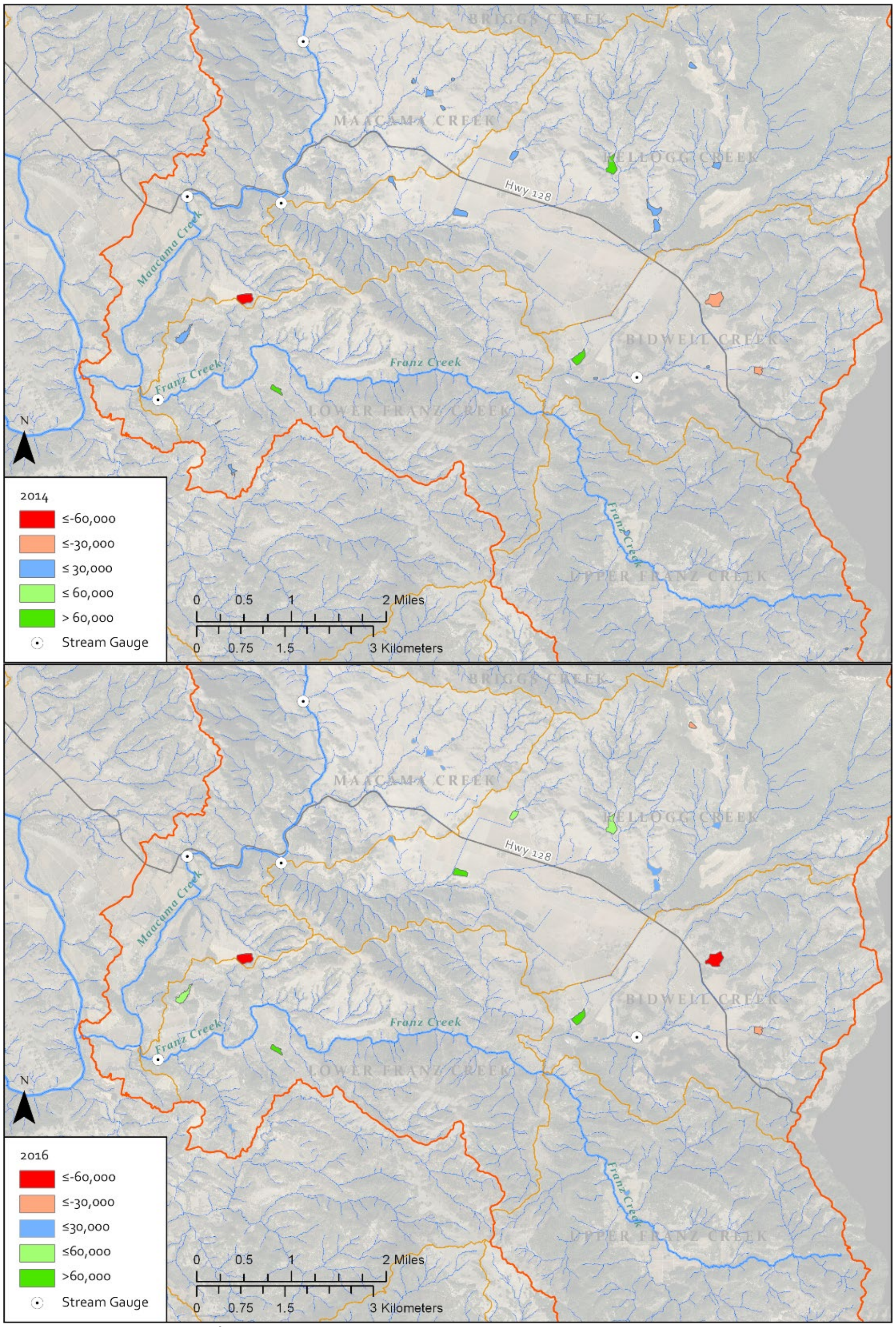


Figure 9: Residual (m^3) comparing EPV & user reported water data for 2014 (top) & 2016 (bottom) water years

3.3 HYDROLOGIC ANALYSIS

3.3.1 DISCHARGE, PRECIPITATION & AIR TEMPERATURE

The calculated 25-year precipitation average was compared to previous estimates by Laurel Marcus & Associates (2004). The Hawkeye RAWS station had a 25-year average of 1055 mm per year. This is similar to historical records near the watershed at the Healdsburg and Healdsburg 2 stations which were 1067 and 1043 mm per year respectively (Laurel Marcus and Associates, 2004).

The 2012-2015 water years are classified as dry-type year, 2016 is classified as an average-type year and 2017 is classified as a wet-type year (*Table 4*). While the dry-type, average-type and wet-type years are classified as such, it is worth noting that there were cumulative patterns of dryness. For example, the 2014 water year not only had well below average precipitation (56% of average) but it followed two years of below average precipitation as well. While the 2016 water year had just slightly below average precipitation (92% of average), it followed 4 dry-type years. The 2017 water year was well above the average annual precipitation rate (168% of average).

Type of Year	Water Year	Total Precip (mm)	% of average	# of Days of Rainfall in WY	# of Days of Rainfall Over Average Rainfall Intensity (14mm)* in WY	# of Days of Rainfall Over Average Maximum Daily Rainfall (98mm)* in WY
Wet-type	2010	1297	123%	-	-	-
Wet-type	2011	1233	117%	-	-	-
Dry-type	**2012**	770	73%	-	-	-
Dry-type	**2013**	919	87%	-	-	-
Dry-type	**2014**	585	56%	56	13	1
Dry-type	**2015**	838	79%	-	-	-
Average-type	**2016**	972	92%	100	25	0
Wet-type	**2017**	1768	168%	105	40	1
**Adopted from Laurel Marcus and Associates (2004)*						

Table 4: Annual precipitation based on 25-year average *(WY = water year)*. Note: - represents data not reported in this table.

When comparing 2014 (dry-type), 2016 (average-type), and 2017 (wet-type) rainfall patterns, 2014 had only 56% of average precipitation and 56 days of precipitation (*Table 4, Figure 10* and *Figure 11)*. There were 13 days where rainfall was over the historic average intensity rainfall of 14mm (Laurel Marcus and Associates, 2004) with only one event occurring over the average maximum daily rainfall for the water year of 98 mm (Laurel Marcus and Associates, 2004).

Discharge patterns have similar patterns of peak flows following rainfall events for all gauges in 2014 (*Figure 10* and *Figure 11)*. The peak flows are also similar between Bidwell and Franz in 2017. There are peak flows that exist at all gauges in 2016 except Redwood during the month of December (see *section 3.3.2*).

Air temperatures approached 0°C on 23 March 2013 and on 6 April 2015 (*Figure 12*). Air temperatures exceeded 32°C on 30 June and 16 August 2015.

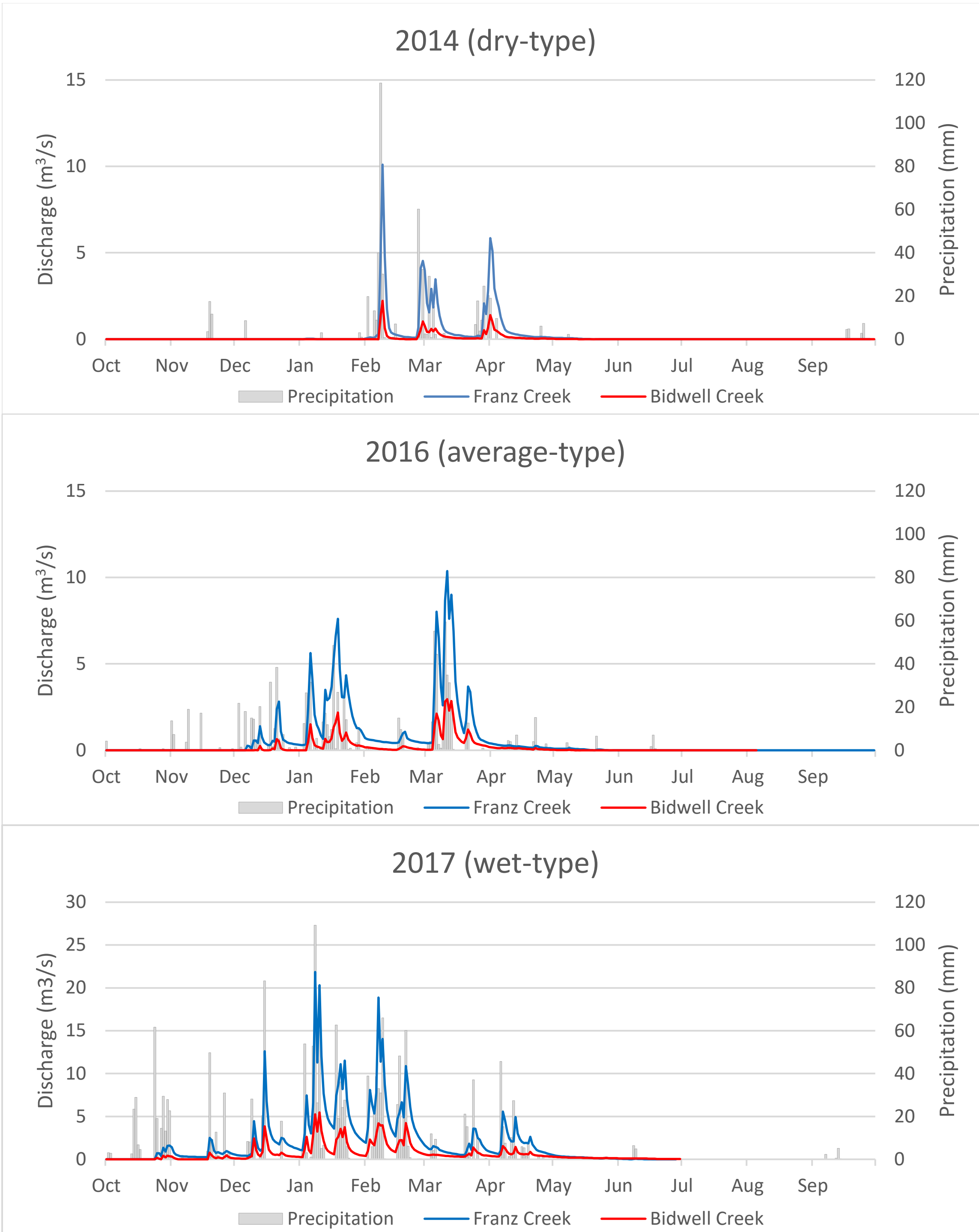


Figure 10: Discharge and precipitation at Franz Creek and Bidwell Creek gauges. Includes 2014 (dry-type), 2016 (average-type) and 2017 (wet-type) water years.

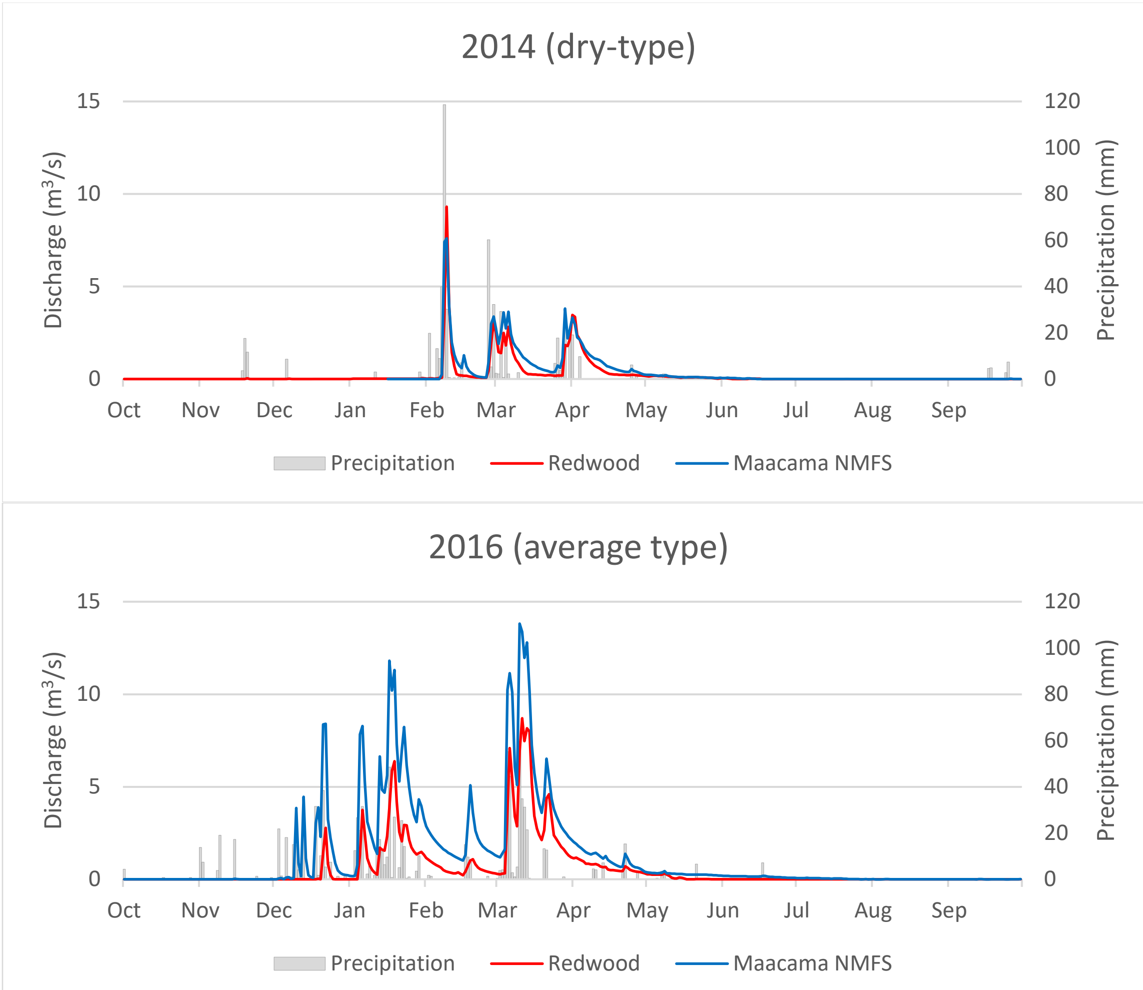


Figure 11: Discharge and precipitation at Redwood Creek & Maacama NMFS gauges. Includes 2014 (dry-type) and 2016 (average-type) water years.

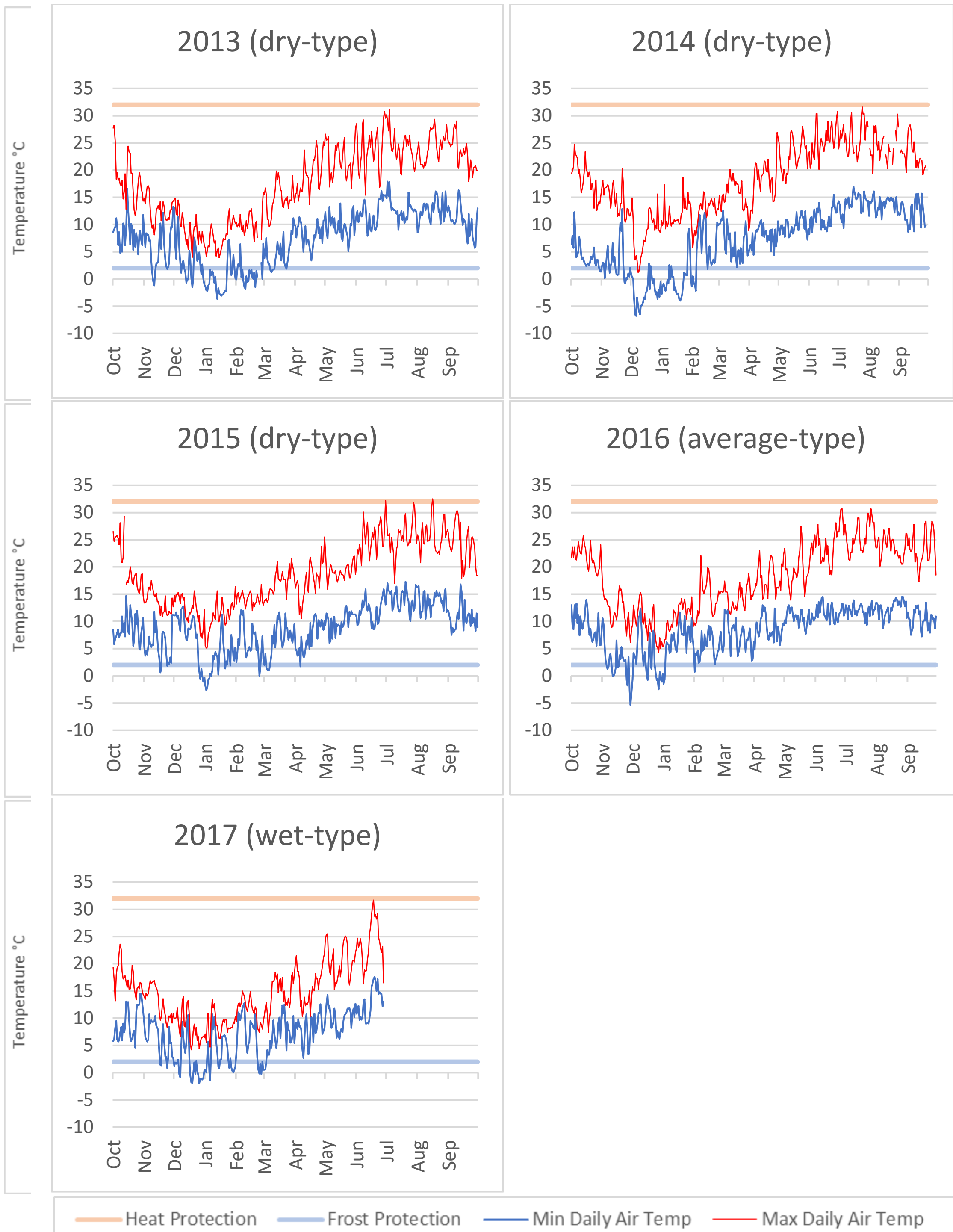


Figure 12: Comparison of maximum and minimum air temp at Franz Creek gauge to typical heat and frost protection thresholds

3.3.2 WATERSHED DISCHARGE IMPAIRED BY PRIVATE RESERVOIRS

The calculated duration of impaired discharge as private reservoirs are filled at the beginning of the rainy season was most extreme in 2014 which has both the most days of impairment and the largest range as well (*Figure 13)*. The range between the minimum and maximum days of impairment was least in 2016. The duration of impairment generally decreases the wetter the year. The exception to this is at Bidwell which has almost an opposite trend with the shortest impairment in 2014.

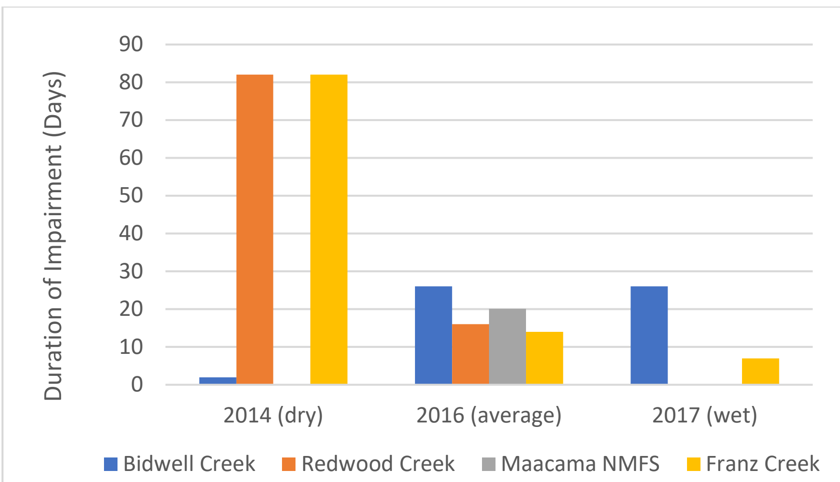


Figure 13: Duration of Impairment for dry-type, average-type and wet-type water years. There was no data for the following: Maacama NMFS 2014, Redwood Creek 2017, Maacama NMFS 2017

Impaired discharge is shown in addition to the late season dry-season intermittency in *Figure 14* and *Figure 15* relative to the hydrograph. Impairment persisted longer into the water year at Bidwell then at Franz during 2012, 2016, and 2017 with impairment ending 50 days, 17 days and 19 days later respectively. Impairment persisted longer into the water year at Redwood than at Maacama NMFS in 2016 (33 days). In addition, both Bidwell (2012) and Redwood (2016) lack peaks of discharge that are present at other gauges.

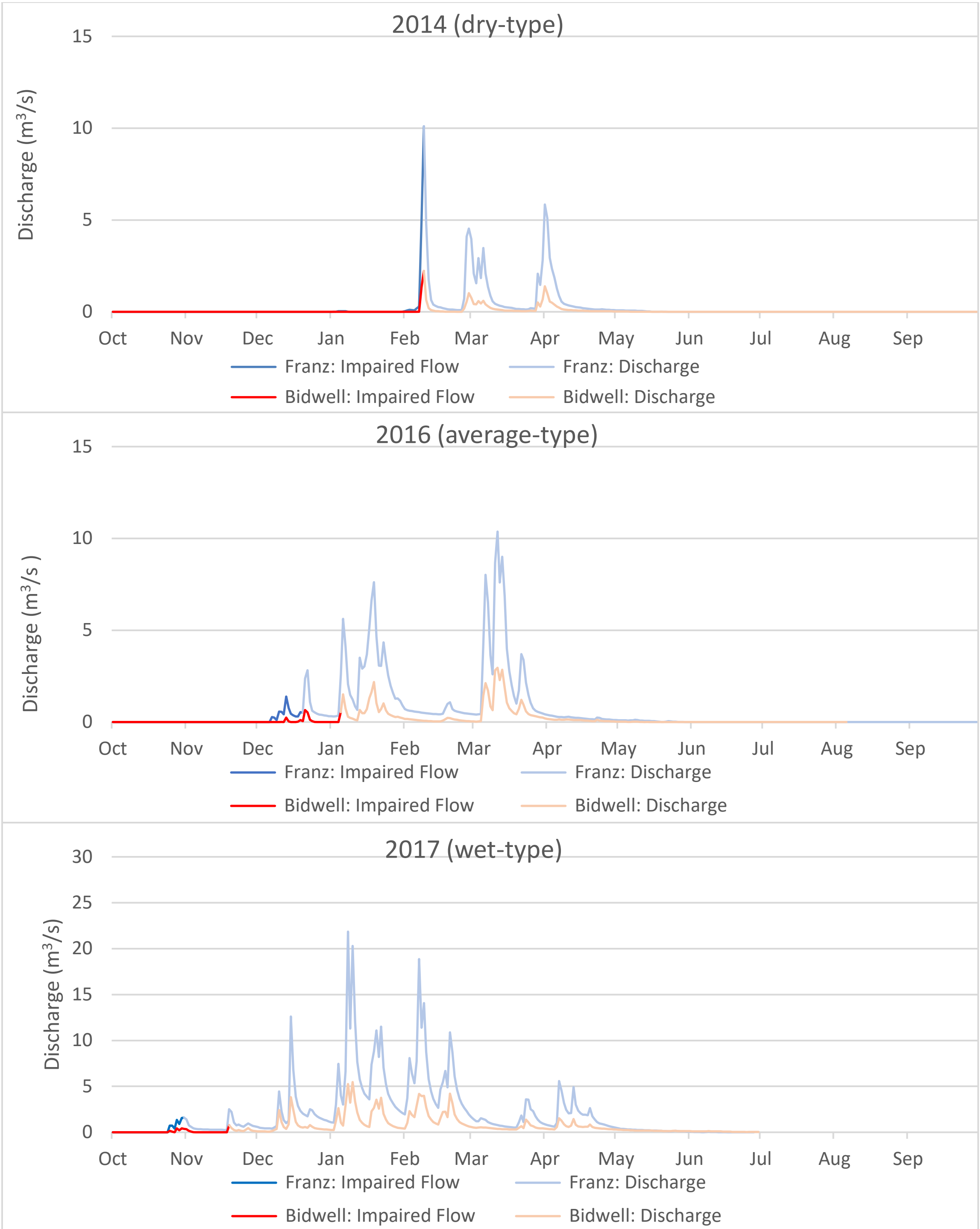


Figure 14: Discharge impaired at Bidwell Creek and Franz Creek gauges by private reservoirs during dry-type (2014), average-type (2016) and wet-type (2017) water years.

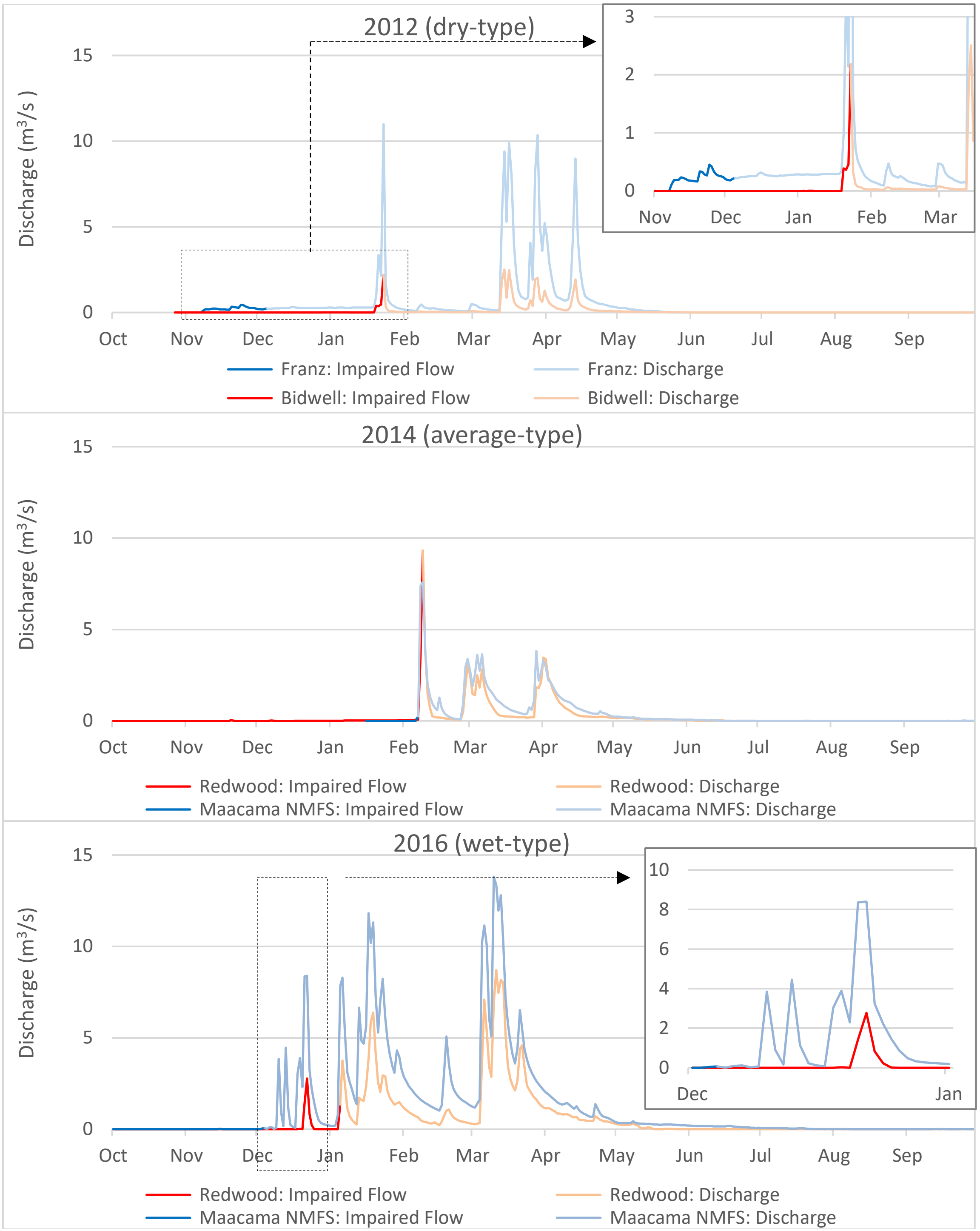


Figure 15: Discharge impaired at Bidwell Creek and Franz Creek Gauges by private reservoirs during a dry-type (2012) year (top). Discharge impaired at Maacama NMFS and Redwood Creek gauges by private reservoirs during a dry-type (2014) and average-type (2016) years (middle & bottom).

3.3.3 STREAM INTERMITTENCY

Bidwell, Maacama USGS, Maacama NMFS and Franz had progressively longer periods of intermittency from 2012-2015 as the dry-type year progressed even if precipitation was higher than the previous year (*Figure 16* and *Table 4*). Redwood also showed progressively longer intermittency duration from 2013-2015. All gauges in the 2016 (average-type) water year had shorter intermittency than any dry-type year. Maacama USGS was perennial in 2017 (wet-type) and is the only gauge with calculated intermittency that year.

Redwood, Bidwell and Franz have higher proportions of EPV to the watershed area above the gauge than do Maacama USGS and Maacama NMFS (see *Table 2*). The duration of intermittency is greater at Redwood, Bidwell and Franz than it is for Maacama USGS and Maacama NMFS every year (*Figure 17*).

Except for Redwood and Bidwell in 2016, intermittency began earlier during dry-type conditions (2012-2015) than during average-type or wet-type conditions (*Figure 18* and *Table 4*). In addition the end of intermittency consistently persisted later into the season following dry-type water years than it did following average-type or wet-type water years (*Figure 18* and *Table 4*).

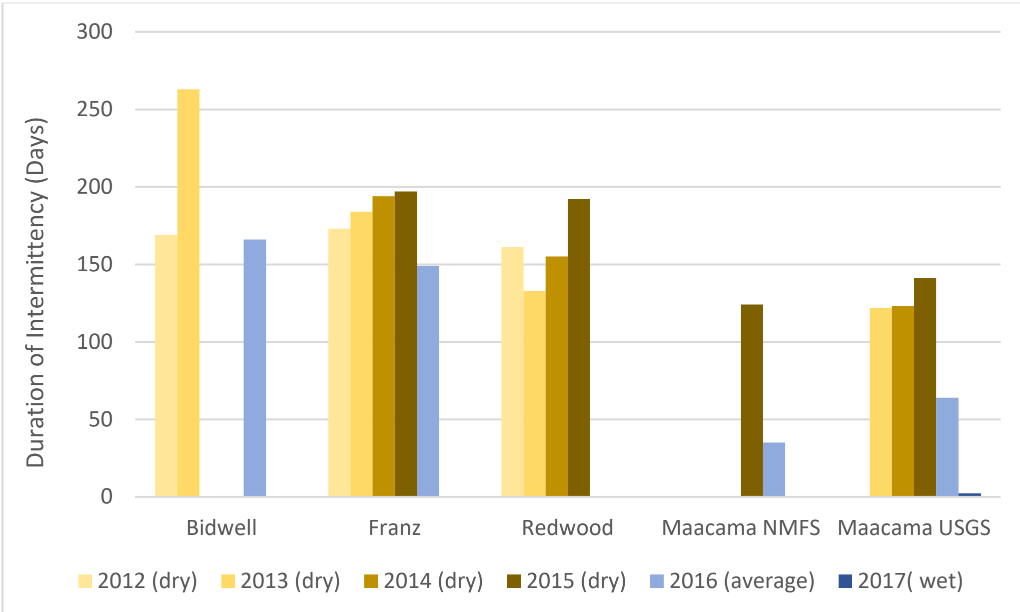


Figure 16: Intermittency duration by stream gauge. Dry-type years shown in browns and yellows. Average-type and wet-type years shown in blues (see *Table 4*). Note: there was no data for the following: Bidwell: 2014, 2015, 2017, Redwood: 2016, 2017, Maacama NMFS: 2012, 2013, 2014, 2017, Franz: 2017, and Maacama USGS: 2012. Maacama USGS was perennial in 2017

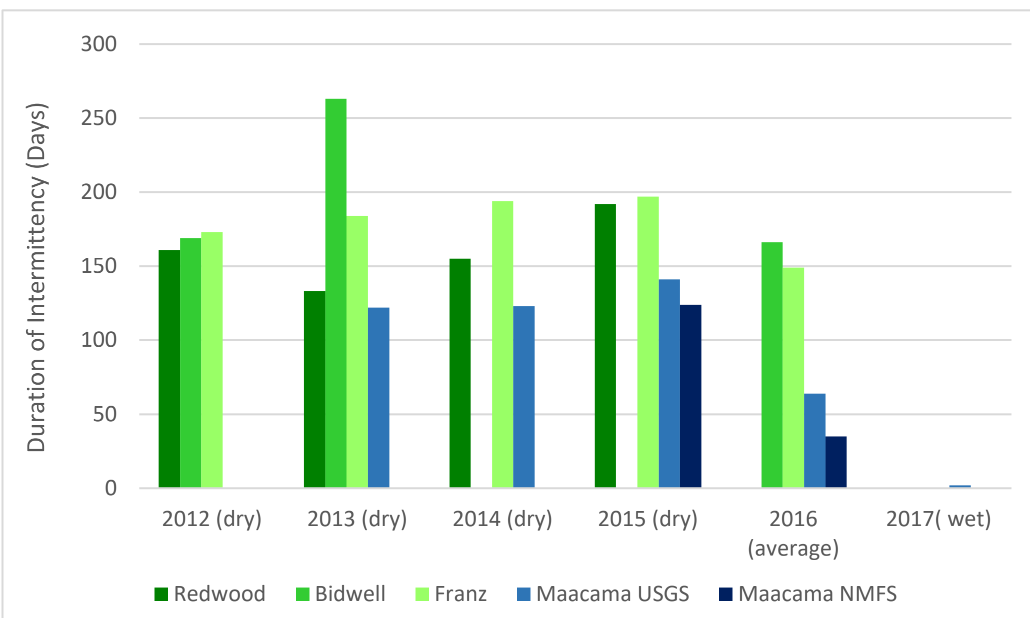


Figure 17: Intermittency duration by water year. Highest proportion of EPV to watershed area shown in green and lowest proportion of EPV to watershed area shown in blue (see *Table 2*). Note: there was no data for the following: Bidwell: 2014, 2015, 2017, Redwood: 2016, 2017, Maacama NMFS: 2012, 2013, 2014, 2017, Franz: 2017, and Maacama USGS: 2012. Maacama USGS was perennial in 2017

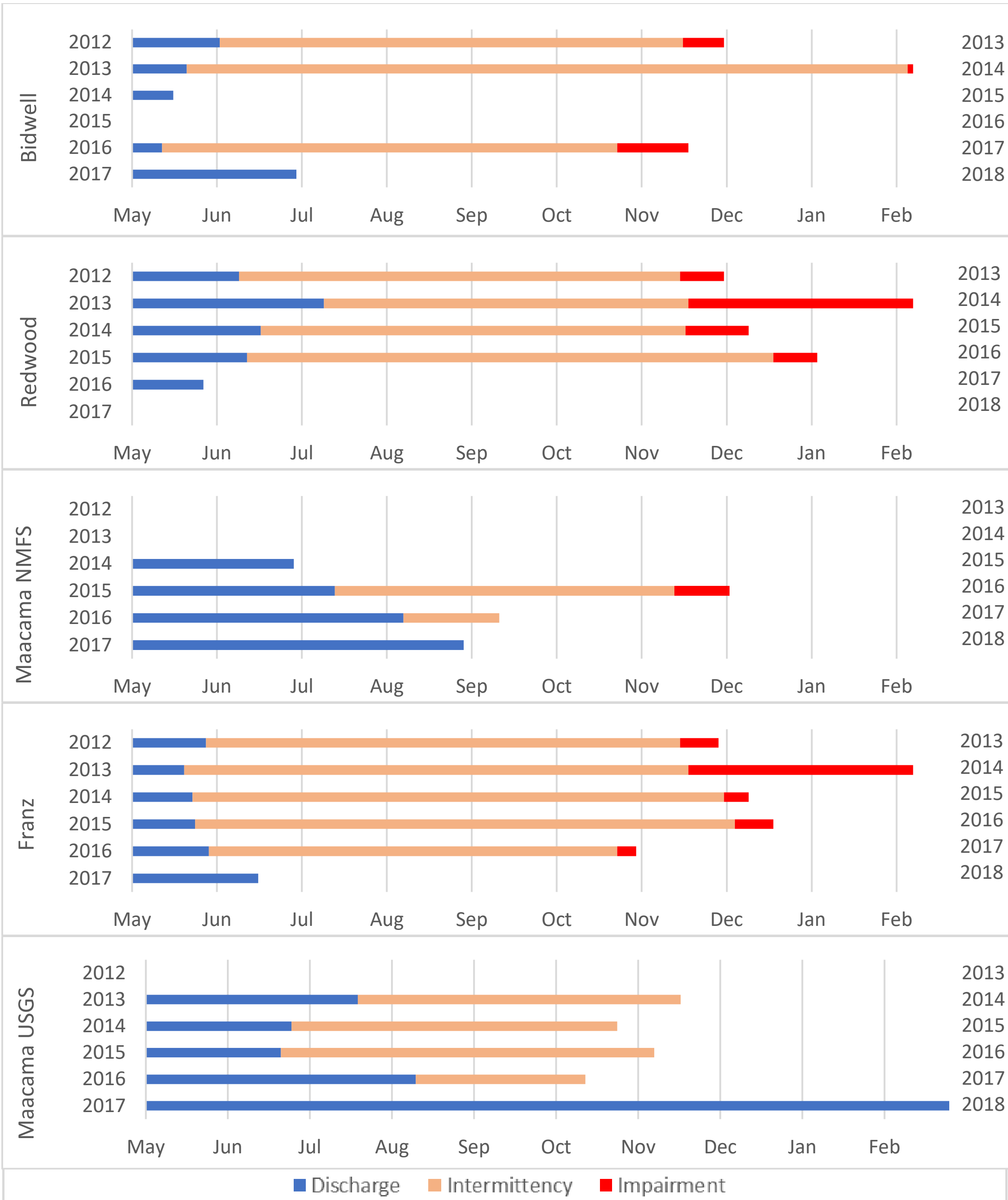


Figure 18: Discharge until the beginning of intermittency, dry season intermittency and the calculated impaired discharge at the beginning of the wet season during the following water year. Water year begins October 1st. Note: if no discharge is shown then there was no data available. No data was available to calculate intermittency for NMFS gauges in 2017, Bidwell 2015, Maacama NMFS 2013, and Maacama USGS 2012. No data was available to calculate impairment for all gauges in 2017 in addition to Maacama NMFS 2016-2017 and Maacama USGS 2013-2018.

3.3.4 WATER TEMPERATURE & LIKELY FISH SURVIVAL

Instream water temperatures measured at the Franz Creek gauge went above the target range of good survival and growth (10-15°C) for coho (NMFS, 2008) all water years (*Figure 19*). The threshold was exceeded in October every year and April-September in 2012-2015 and May-September in 2016. In 2017 the threshold was exceeded October-November along with March-June. Water temperature data were available past June 28, 2017.

The daily maximum temperature was below the good survival and growth target (10°C) for coho (NMFS, 2008) for all water years except 2015 (*Figure 19*). Temperatures were below the threshold December-February in 2012, November-March in 2013, October-February in 2014, November in 2015, November-January in 2016, and December-June 2017.

The threshold that indicates a major decline in the likelihood of survival (18°C) for coho (NMFS, 2008) was reached every year. The threshold was exceeded June-August in 2012, June-September in 2013, May-September in 2014, July-August in 2015, June-July in 2016 and April-June in 2017.

Water temperatures were well above the preferred water temperature (14.4°C) for steelhead (NMFS, 2008) in the 2012-2017 water years (*Figure 20*). The threshold was exceeded in October all years. It was also exceeded April-September in 2012, December and April-September in 2013, April-September 2015, April-September 2016 and November and March-June in 2017.

Water temperatures were below the preferred water temperature (7.2°C) for steelhead (NMFS, 2008) in 2012, 2013 and 2014 and was just slightly below that range briefly in 2017. The temperature went below the threshold in December-January 2012, in December-January in 2013, October-February in 2014 and on December 30th in 2017.

In 2013 and 2017 temperatures reached or exceeded the lethal limit (23.9°C) for steelhead (NMFS, 2008). This occurred June-July of 2013 and June of 2017.

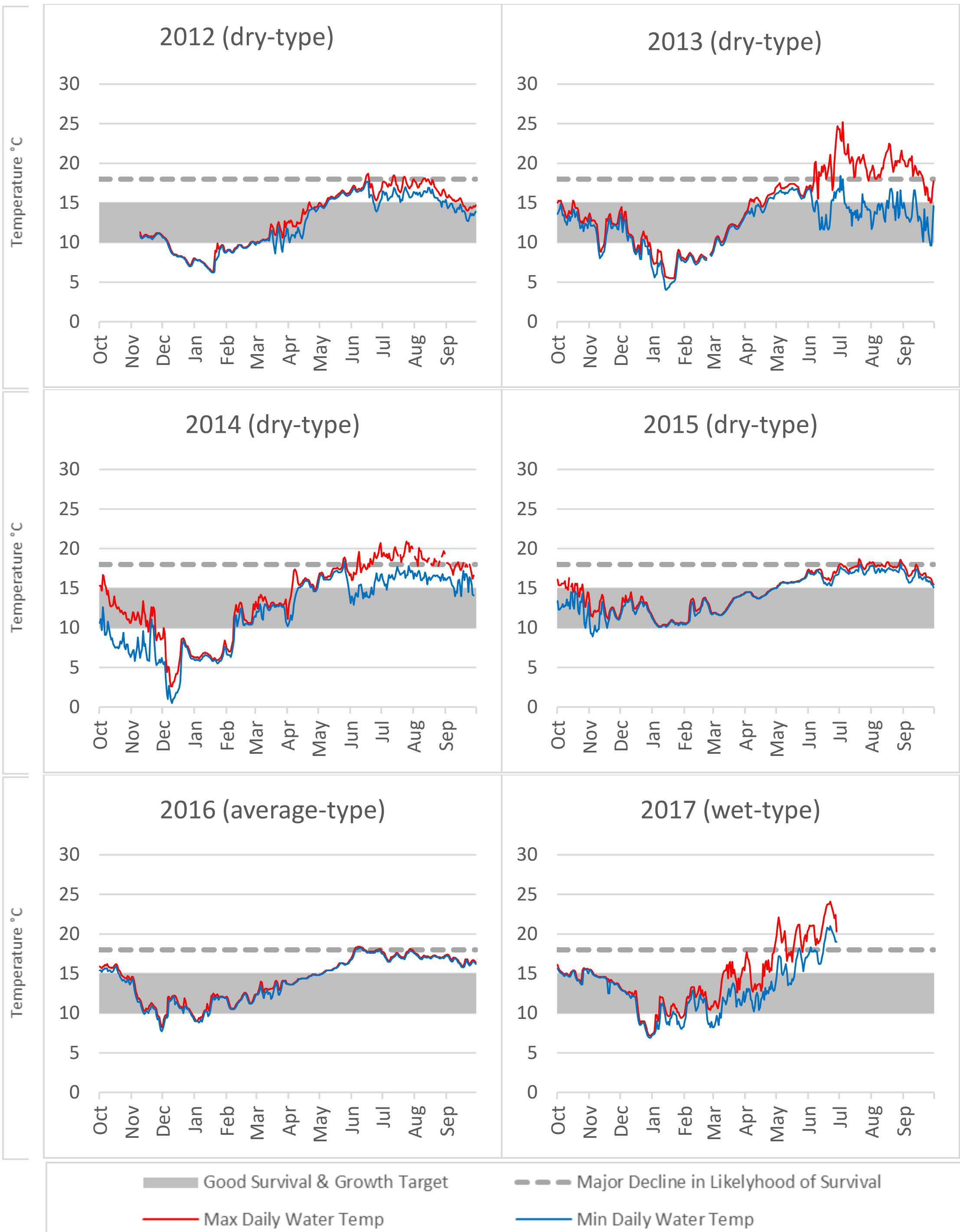


Figure 19: In situ measured water temperature vs. target temperatures for coho at the Franz Creek gauge.

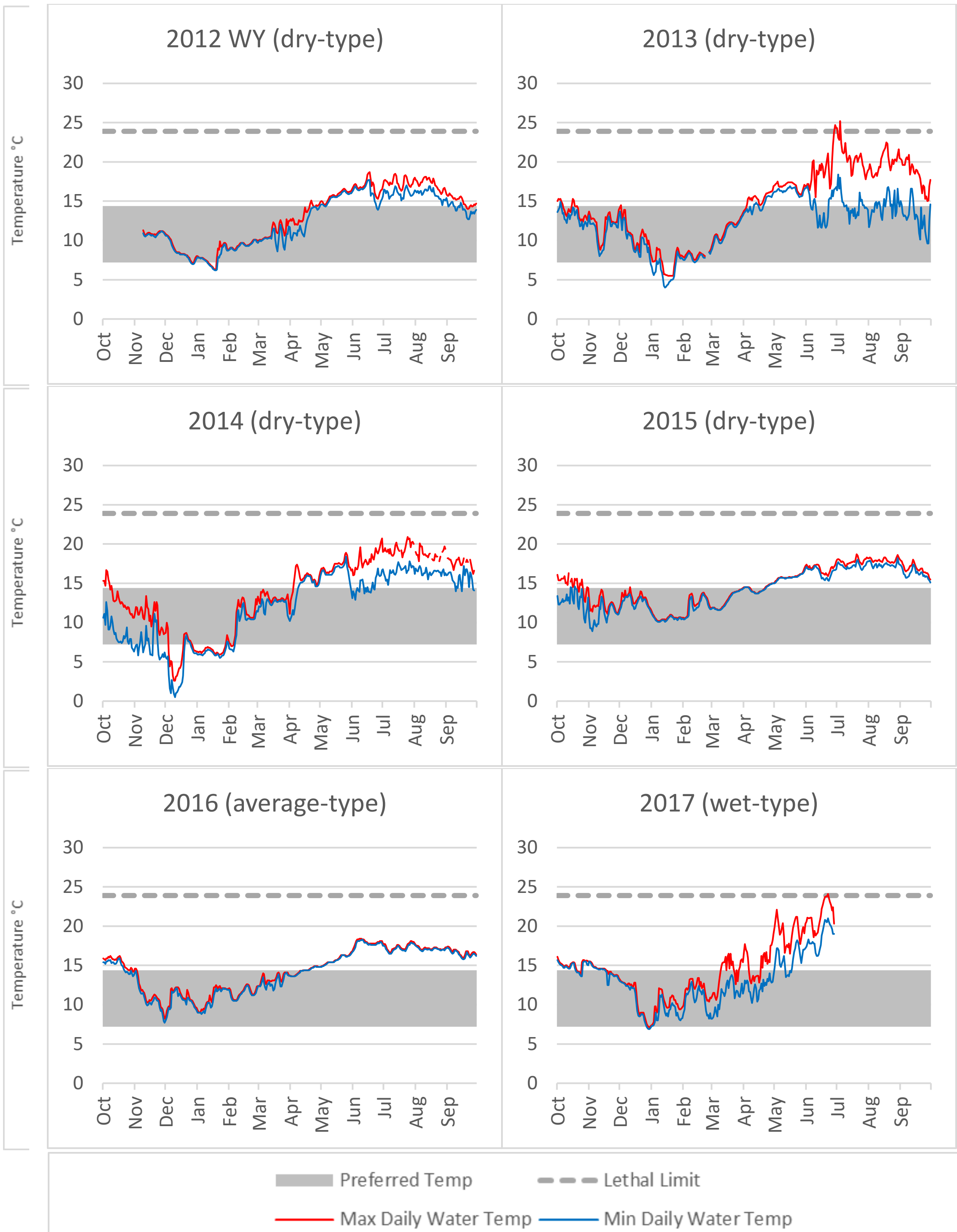


Figure 20: In situ measured water temperature vs. target temperatures for steelhead at the Franz Creek gauge.

4.0 DISCUSSION

4.1 COMPARISON OF EPV TO REPORTED WATER USE

EPV is a valuable method that can be used to model impacts to streamflow where reporting is unavailable for reservoir storage. Deitch (2006) suggests that impacts to streamflow need to be modeled as demand rather than need by reflecting the water that can be diverted from the stream at any specific time. Previous studies have modeled estimated water use based on land use (Merenlender et al., 2008), including modeling hypothetical reservoirs to investigate potential spatial variability (Grantham, 2010). Grantham et al. (2014) digitally mapped reservoirs from aerial photography; however they had a subset of reservoirs that had a known volume that was used to create an empirical relationship between surface area and volume. This study builds upon that research by developing and applying the EPV method to estimate water used in the Maacama watershed.

In order to compare user reported data for both years, ponds with an EPV were matched to an eWRIMS ID. The location of the diversion as per eWRIMS GIS data was not always at the pond so consequentially only 44% of the ponds were able to be matched with eWRIMS data. Previous work by Deitch et al. (2016) uses water rights records to investigate water use in the state. They use the permitted face amount, which is the maximum allocated amount of water allowed through that permit and is similar to the process used by the State Water Board which is used for determining water availability to inform policy and water rights allocations (Deitch et al., 2016). Grantham & Viers (2014) found that water rights associated with appropriative rights were commonly over estimated. When compared to the EPV, the face amount is about 60% higher which is in agreement with Grantham & Viers (2014).

User reporting during 2014 (dry-type) was approximately 30% more than the EPV, indicating that the EPV approach might be underestimating storage within the reported ponds. The 2016 (average-type) year had about 20% more water use reported than the EPV, which is reasonable that there would be less water use reported during a year with higher precipitation.

There was a significant relationship between the EPV and landowner reported water use for 2014 ($p\text{-value}<0.01$) and 2016 ($p\text{-value}<0.01$). Both 2014 and 2016 were statistically significant; however the r-squared value shows a stronger correlation in 2014 ($R\text{-squared}=0.54$) than in 2016 ($R\text{-squared}=0.38$) which is reasonable since the EPV is assumed to be the maximum amount of water stored annually.

The EPV is a useful approach that approximates reservoir surface area and more importantly can be used as a remote technique to estimate the spatial distribution of reservoir storage volume within a watershed. This approach can provide a systematic estimation of the change in volume of the annual water storage in a watershed when there is no known reservoir volume data or public reporting of water use is unavailable. It could also help determine water use in areas with unpermitted water storage. The change in volume may be more important than the capacity of the reservoirs as a metric for policy and watershed management, as it represents the annual amount of water diverted from the watershed. The significant relationship between user reported water use and EPV ($p\text{-value}<0.01$) suggests that EPV could be a valuable tool for estimating water storage in a decentralized watershed management regime where reporting is unavailable.

LIMITATIONS

There are a few limitations to the EPV approach. The first is with cattle paths which could have been misinterpreted as the maximum water line since they also tend to run parallel to the water's edge. Another limitation is the thick vegetation cover in part of the watershed, which prevented some ponds from being able to fully be digitized as both an inner and outer pond. The thick vegetation cover also could have resulted in some estimation of the pond edge, may have affected the LiDAR based DEM or could have potentially caused a disproportionate number of ponds in low lying and highly vegetated areas to be excluded. Evapotranspiration which can be influenced by vegetation cover and pond exposure was also not included in this analysis. In addition, ponds that were easier to identify, digitize and quantify were more likely to be included in the analysis. Finally, it was assumed that the pond water levels were at their lowest level at the time the LiDAR data was collected because it was the peak of the dry season during a dry-type year.

This study did not address the potential impacts of groundwater use on instream flow as data access and uncertainty can be a limiting factor. However, some landowners reported using groundwater during the 2014 and 2016 years (SWRCB, 2015) and it is potentially an important component of instream flow in the watershed during certain times of the year at some locations. There is also potential that groundwater could be used to fill the ponds.

There may be additional ponds that have eWRIMS reported data that were not included in this research. Spatially matching the ponds with EPV with eWRIMS GIS data was not always clear because the location of some storage ponds are shown at the riparian point of diversion and can be a fair distance from the actual ponds.

4.2 INSTREAM DISCHARGE ANALYSIS

4.2.1 INTERMITTENCY

DURATION

Streamflow in the Maacama watershed typically recedes in the spring and by the end of the summer can be near or reach intermittency (Grantham et al., 2014); however, acceleration of intermittency in the Maacama watershed has been shown to occur due to surface water diversions in the dry season (Deitch, 2006; Deitch et al., 2009a). Previous modeling done in the Russian River watershed found that water demands for human use exceeded instream flow by early June (Deitch et al., 2009b). Accelerating the dry season has the potential to increase the duration of intermittency or the intensity of low flows (Grantham et al., 2012). Instream flow is critical for coho and steelhead that are present in the Maacama watershed (CDFG, 2005) as intermittency timing and duration is important to maintaining adequate conditions for specific salmonid life stages (Deitch, 2006; Deitch et al., 2009a).

There was a pattern of progressively longer intermittency from 2013-2015 as the dry-type years progressed and conversely shorter or no intermittency for the average-type and dry-type water years (see *Figure 16* and *Table 4*). Bidwell, Maacama USGS, Maacama NMFS and Franz all had progressively longer periods of intermittency from 2012-2015 and as did Redwood from 2013-2015. This did not directly correlate with precipitation patterns as rates varied from 2012-2015, with 2014 receiving the lowest total rainfall in that period.

When comparing data from this study to historical data at Maacama USGS (*Figure 21*) the duration of intermittency is much higher (≥141 days) than it was historically (<90 days). Intermittency occurred more frequently as well (4 out of 5 years) than it did historically (6 out of 20 years).

The exception to the pattern was at Redwood where intermittency was significantly longer in 2012 than in 2013. Precipitation alone doesn't explain this pattern as 2012 followed two wet-type years and 2014 had even less precipitation (see *Figure 16*).

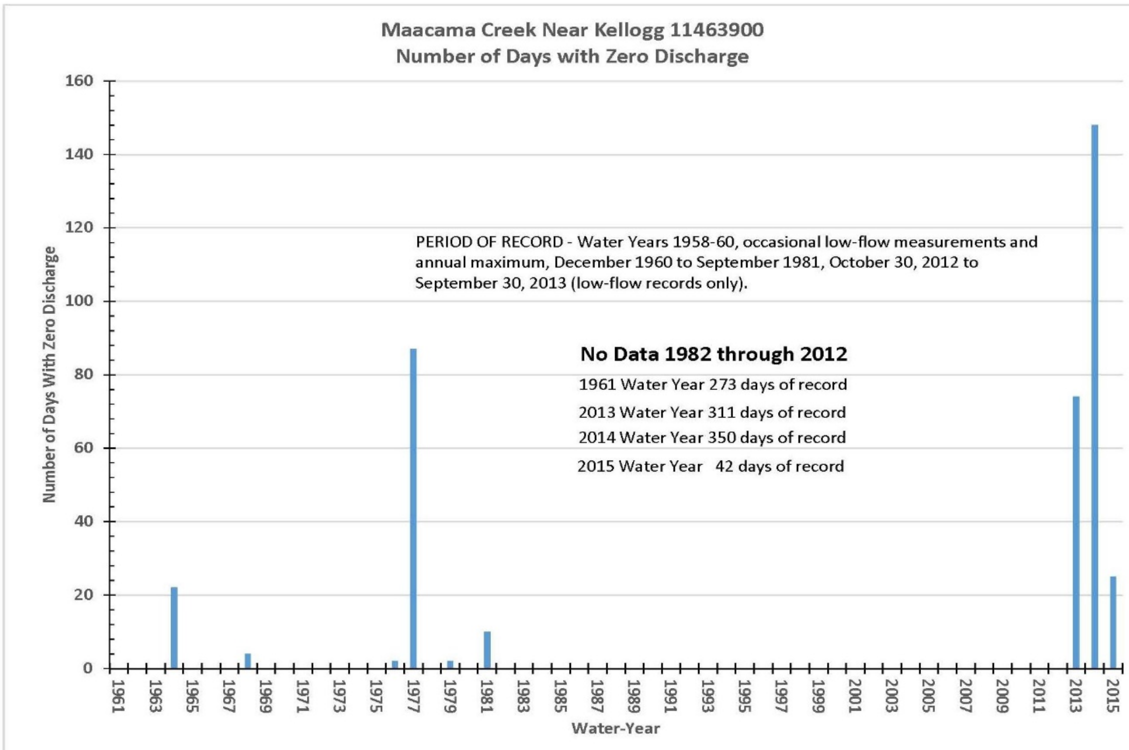


Figure 21: Maacama Creek: historical days of zero discharge (RRISRP, 2016, p. 384)

LOCATION IN THE DRAINAGE NETWORK

Previous modeling done in the Maacama watershed found that the potential for conflict between water security and protecting adequate flow in the Maacama watershed was highest in the upper drainage network (Grantham et al., 2014). Smaller streams tended to have less water available and a significantly lower number of bypass days for adult salmonid migration to begin with (Grantham et al., 2014). Small instream diversions in the Franz creek subwatershed were also found to disproportionally affect the upstream drainage network (Deitch et al., 2009a).

Subwatersheds higher in the drainage network generally appeared to have longer intermittency than downstream in the drainage network (see *Figure 16* and *Table 2*). Bidwell, which is higher than Franz, had longer intermittency than Franz in 2013 & 2016

(79 days and 17 days respectively). At the Redwood gauge, which is higher than Maacama USGS, intermittency is also longer (by 11-51 days).

There are a few exceptions. Intermittency at Bidwell in 2012 was 4 days shorter than at Franz. In addition, Maacama NMFS has shorter intermittency than at Maacama USGS, despite being higher in the drainage network. The shorter intermittency could be because there is less impairment above Maacama NMFS than at Maacama USGS (see *Table 2*).

PROPORTION OF STORAGE TO WATERSHED AREA

In general the gauges with the higher proportion of EPV to watershed area had longer periods of intermittency (see *Figure 17* and *Table 2*). Bidwell and Franz, which have higher proportions of EPV to watershed area, have longer durations of intermittency than Maacama USGS and Maacama NMFS every year. While Redwood always has longer intermittency than Maacama USGS and Maacama NMFS, intermittency at Redwood is always shorter than intermittency at Bidwell and Franz for the corresponding year. This is notable as Redwood has a much higher percentage of EPV to watershed area than all the other gauges (see *Table 2*). The Redwood gauge is below an area of high agricultural use (Deitch et al., 2009a) and also the Knights Valley groundwater basin (see *Figure 5*) so there is potential that these two factors might play into the discrepancy between the trends occurring at Redwood and the trends occurring at all of the other gauges. This is difficult, though, to confirm without having additional accurate water use reporting for the entire watershed.

PATTERNS OF TIMING

In general intermittency began earlier during dry-type conditions (2012-2015) than during average-type or wet-type conditions (see *Table 4* and *Figure 18*). There were some exceptions, for example Redwood appears to have a somewhat opposite trend than the other stream gauges with intermittency beginning earliest in 2016 (average-type), even though that year had more precipitation than any of the dry-type years. In addition, intermittency began earlier in 2012 than 2014. Precipitation does not explain this observation as not only was there more precipitation in 2012 but also 2012 followed wet-type years, while 2014 followed dry-type conditions. There is concentrated vineyard development above Redwood Creek (see *Figure 5*) (Deitch et al., 2009a). Further investigation is needed to determine if there are any other anthropogenic controls that are influencing the start of intermittency in the stream such as deep groundwater pumping or groundwater recharge that may not otherwise be expected at that gauge during dry-type years.

Another exception is at Bidwell where intermittency starts earliest during 2016 (average-type), which is earlier than 2012-2014 (dry-type) (see *Table 4* and *Figure 18*). The 2016 year had just slightly below average rainfall and followed multiple years of below average precipitation; however, Bidwell and Redwood were the only gauges to show this trend with the average-type year having the earliest start of intermittency.

While the start of intermittency was not always earliest during dry-type years, the end of intermittency consistently persisted later into the season following dry-type water years than it did following average-type or wet-type water years (see *Table 4* and *Figure 18*). Summer rearing habitat in some coastal streams is potentially as important as winter habitat as a limiting factor for the survival of juvenile salmonids (Grantham et al., 2012). Reductions in streamflow during low-flow conditions, whether from diversions for human use or drought, is likely to threaten the survival of juvenile salmonids (Grantham et al., 2012). Cumulative years of dry-type conditions brought on longer durations of intermittency and led to the end of intermittency persisting later into the water year (see *Figure 18*).

There can be tradeoffs between protecting instream flow needed for adequate environmental protections and water security for agriculture particularly during dry-type years (Grantham et al., 2014). Because of extreme low flow conditions during the summer it is important for policies to concentrate water withdrawals during the winter rather than in the summer (Deitch et al., 2016). This has important policy and management implications. For example, policy must reflect the changing scenarios of drought if private reservoirs in the Maacama watershed are to reduce impacts to instream flow during low flow conditions including the end of the dry season. While new appropriative water rights restrict diversions to begin after December 15th (Deitch et al., 2016), the policy only first became adapted in 2010 and does not include water rights granted before 2010 (SWRCB, 2019b). In another coastal California watershed the restriction was found to only include a small fraction of all diversions (Deitch et al., 2016). The state of California does not have accurate accounting of how much water most water rights holders use (Grantham & Viers, 2014). Water policy that can manage and regulate all water users will likely be useful in establishing adequate ecological protections. In addition, exacerbated dry conditions due to climate change and cumulative years of drought need to be considered when determining policies that regulate the timing reservoirs are refilled.

In this study, often intermittency didn't end until mid-November, but perhaps more importantly there were years when intermittency didn't end until mid-January or even February as is the case in 2013 at Bidwell (see *Figure 18*). Monitoring of instream flow at the onset of the rainy season to determine if salmonids have adequate conditions for migration and spawning is warranted as there is potential that diversions to refill ponds could overlap at key times of the year. Flow policies that maintain minimum flows and are adaptive in response (Palmer et al., 2009) particularly during drought conditions are important so that streamflow restoration is not hindered at the end of the dry season. Funding and political will are essential to improving policy, monitoring efforts and the enforcement (Grantham & Viers, 2014) necessary to follow through with instream flow restoration in a decentralized management regime.

FROST AND HEAT PROTECTION

In the Maacama watershed, the effects of small instream diversions were found to impact the watershed disproportionally, more greatly impacting local hydrology along with the upstream drainage network (Deitch et al., 2009a). Deitch et al. (2009) found that when temperatures approached 0°C or exceeded 32°C in the Maacama watershed, streamflow receded due to instream water diversions used for frost and heat protection, respectively. Air temperatures approached 0°C in 2013 and 2015 (see *Figure 10*) potentially causing egg mortality, loss of food supply and contributing to early dry season intermittency (Deitch et al., 2009). In addition, air temperatures reached 32°C twice in 2015.

In this study, the earliest start of intermittency at Franz occurred in 2013 and at Maacama USGS in 2015, each following one of the years of potential heat or frost protection (see *Figure 12* and *Figure 18*) similar to previous work (Deitch et al., 2009a). While both 2013 and 2015 had below average precipitation, the precipitation in 2012 and 2014 was even lower so early intermittency can't be explained by precipitation rates alone (see *Table 4*). Deitch (2009a) found that frost protection during periods of base flows had the potential to accelerate intermittency by up to 2 months in the Maacama watershed. Conversely, the latest end of intermittency occurred following either 2013 or 2015 for every gauge, which raises questions about why prolonged intermittency would occur those years and whether that could be a result of frost protection.

Heat protection was likely on 30 June and 16 August 2015 as air temperatures exceeded 32°C. The latest end to intermittency following the 2015 water year occurred at Redwood, Maacama NMFS and Franz gauges. It's difficult to determine whether this was directly related to heat protection with the available data. It's possible that intermittency was accelerated due to instream diversions during low flow conditions similar to what Deitch observed with frost protection (Deitch et al., 2009a). An alternate explanation for the late end to intermittency in 2015 could be because 2015 was the fourth cumulative dry-type year.

4.2.2 IMPAIRMENT

LOCATION WITHIN THE DRAINAGE NETWORK

Calculated impairment ended later at gauges located higher in the drainage network (see *Table 2, Figure 14, Figure 15,* and *Figure 18*). For example, impairment at Bidwell ended either the same day or later in the year than at Franz. This is consistent with previous work that found effects of diversions are highest upstream in the drainage network (Deitch et al., 2009a; Grantham et al., 2014). In addition, there are a few examples where there is no discharge at sites that are higher in the drainage network and are not observed as expected in the hydrograph based on other stream gauges (see *Figure 15*). In 2016, Redwood lacked multiple discharge peaks which were observed at Maacama NMFS, Franz and Bidwell as well. Discharge at Redwood is generally greater or equal to Bidwell the remainder of the water year. In 2012, something similar can be observed at Bidwell as well. Although local storm events can affect Maacama and Franz subwatersheds differently (RRISRP, 2016), this result does not seem consistent with discharge quantities and patterns throughout the rest of the year.

PRECIPITATION TIMING AND INTENSITY

Dry-type conditions led to the most extreme impairment. In general, when comparing a dry-type (2014), average-type (2016), and wet-type (2017) water year dry-type conditions had the longest impairment, while the wet-type year had the shortest impairments as expected (see *Figure 13* and *Table 4*). The exception, however, is at Bidwell which had the opposite trend of impairment duration with 2014 being the shortest duration of impairment. Perhaps this is because Bidwell is located higher in the drainage network than the rest of the gauges or because it is already more affected by dry-type conditions and therefore there is no flow recorded at the gauge until the highly episodic rainfall event in February 2014 (see *Figure 10, Figure 11* and *Table 2*).

The timing and intensity of precipitation contributed to the timing of the end intermittency beyond just the overall amount of precipitation. The majority of the rainfall in 2014 occurred late in the season (starting in February) resulting in some of the longest impairments (82 days) and the latest end to impairment (2/10) (see *Figure 13, Figure 10, Figure 11,* and *Figure 18*). In terms of precipitation the 2014 and 2017 water years were the most extreme as not only the driest and wettest years but they also both had large episodic rainfall events. The 2014 & 2017 water years also had the greatest range in the duration of impairment (80 days & 19 days respectively) suggesting that impairment could also affect the watershed unevenly during extreme climatic conditions (see *Figure 13* and *Figure 10*). This has implications for future management considerations as climate change is expected to exacerbate natural climate variability in coastal California (NCRP, 2014).

The "fill-and-spill" method was used when calculating impairment. Timing, duration and intensity of precipitation along with the timing of reservoir replenishment are important factors in influencing the impacts of reservoir storage on instream flows. According to the Russian River Independent Science Review Panel report (2016) new water rights policies restrict winter diversions to occur only between 12/15 through 3/31; this however, only applies to new permits and is not retroactive. There are water rights holders that reported diverting water in the watershed as early in the year as November (SWRCB, 2015). The current policy would be more effective in protecting instream flows if it applied to all diversions in the watershed without any exceptions.

There may be more water being diverted for private reservoir storage than is reflected in the estimated EPV value (see *section 4.1*), which could lead to even longer durations of impairment extending the end of intermittency further into the water year than this study suggests. Deitch (2009a) found that small instream diversions during periods of base flow had the potential to accelerate intermittency by up to 2 months in some cases resulting in low flow conditions both earlier in the season and for a longer duration of time. The potential for diversions early in the rainy season to reduce instream flow until peak flows may be dependent on the timing of diversions in combination with the timing, duration, and intensity of rainfall during the rainy season.

Managers may begin to consider whether diversions to refill private reservoirs are extending the end of intermittency or reducing instream flows at the onset of the rainy season that might prevent migration or hinder other salmonid life stages. The calculated impairment in this study ranged from 2-82 days (see *Figure 18*). The peak coho migration occurs in December and January with spawning occurring soon after the adults reach spawning sites (NMFS, 2008). Multiple years of dry-type conditions in Franz creek shows a progressively later end to intermittency which by 2015 ends as late as December (see *Figure 18*). This is only the end of intermittency (see *Figure 18*) and is not a reflection of adequate flow needed for migration, spawning and other life stages necessary to the year round needs of both coho and steelhead rearing. Migration during dry-type years or years where rainfall occurs late into the year could impair discharge during periods of migration and spawning. Additional monitoring could help determine changes in flow patterns that affect winter migration and spawning patterns. If intermittency is extended it has the potential to put additional stress on rearing salmonids already in the stream as they wait for flow to be restored (Grantham et al., 2012).

The timing of storage diversions can affect whether reservoirs reduce or add to stress on aquatic life during low-flow conditions. Climate change will exacerbate current risks and managers will need to respond at a watershed level to ensure adequate environmental flows (Palmer et al., 2009). Small-scale diversions in a decentralized watershed management regime have the potential to alleviate some of the dramatic effects that occur with large-scale water projects (Grantham et al., 2014). However, the success of these approaches will depend on the timing of diversions and the implementation over the entire watershed.

Successful watershed management in the Russian River watershed relies on environmental science professionals, the public sector and all stakeholders to work together towards a common goal. This positive approach will help propel the changes necessary to maintain adequate streamflow in the watershed. Decentralized watershed management is more complicated than large-scale water projects and involves not just public and private input but also follow-through by many different stakeholders to put policies into practice to protect ecosystems. This may be challenging, however, due to state policies that grandfather in water rights that are neither able to be monitored or altered to respond to the daily requirements of aquatic organisms.

LIMITATIONS

The water balance equation developed by Deitch et al. (2009b) assumes that the discharge recorded at the stream gauge (I) is unimpaired by existing reservoir storage when in fact discharge at the stream gauge already reflects current water storage diversions. This study further develops the comparison of impaired discharge due to refilling private reservoirs by investigating spatial variability within the watershed by including data from multiple stream gauges that are dispersed throughout the watershed.

Limitations to data availability occurred with both streamflow and precipitation data. All stream flow gauges are managed for low flow conditions. The NMFS discharge data is calculated using flow curves to extrapolate high flow conditions and is therefore an estimation of instream flow. In addition, some stream gauge data was incomplete for the water year. The 2017 Maacama NMFS gauge recorded having a bent staff plate even though discharge data was still available. Finally, the USGS gauge is only managed for low flow conditions (≤200 cfs) (A. Watson, written communication, February 25, 2019) which results in data gaps and limits its usefulness for impairment calculations.

When calculating impairment, the 'fill-and-spill' approach was assumed to start at the beginning of the water year even though new policy limits the winter diversion season from 12/15-3/31. It is also assumed that all stored water derives from surface flow or precipitation rather than being pumped from instream flow or ground water. This method assumes that all streamflow at the gauge was filling the EPV above that gauge until the total EPV had been filled regardless of location within the watershed and does not account for evapotranspiration.

4.3 WATER TEMPERATURE & LIKELY FISH SURVIVAL

Instream water temperature is important for coho survival and is coupled with instream flow during the dry season (Grantham et al., 2012). Maintaining adequate habitat suitability all year is important, as rearing can occur in the stream for just over a year for coho and for two or more years in the case of steelhead (NMFS, 2008; NOAA, 2012). Growth is perhaps the most important metric for determining fish fitness (Myrick & Cech, 2004).

Water temperatures at the Franz Creek gauge exceeded ideal conditions for both coho and steelhead growth every year regardless of being a dry-type or wet-type year. It's unlikely that rearing coho survived through the 2012-2017 water years or that rearing steelhead survived through 2013 and 2017 based on temperature data.

Dry-type years exhibited lower maximum water temperatures and wet-type years exhibited higher water temperatures. The 2017 and 2013 years had the highest maximum temperatures (24.1°C and 25.2°C respectively) (see *Table 4*, *Figure 19* and *Figure 20).* There have been instream water temperature trends observed in other tributaries to the Russian River indicating higher water temperatures during wet-type years and lower water temperatures during dry-type years (M. Obedzinski, personal communication, March 15, 2019). Scientists in the area hypothesize that the relative contribution of groundwater to the stream is greater during dry-type years which helps to maintain lower instream temperatures (M. Obedzinski, personal communication, March 15, 2019). This hypothesis also holds true for 2012 and 2015 dry-type years which had low water temperatures (8.7°C and 17.8°C respectively).

The exception to this is observed in 2014. The 2014 water year was the driest year in this study but has a maximum water temperature (20.7°C) which is higher than 2012 and 2016 (see *Table 4, Figure 19* and *Figure 20*). Both 2012 and 2016, however, would be expected to have higher water temperatures based on precipitation data alone. It is possible that the temperature measured that year had some other factor that makes it difficult to compare it to temperature data other years. For example, a different pool may have been used to measure water temperature that year, or perhaps vegetation cover was different above either the pool or the data logger that resulted in higher temperatures.

More study is needed to determine the mechanisms that drive variations in water temperature and the influence groundwater has on instream water temperature. Accurate information about groundwater pumping practices is needed to research if groundwater withdrawals and its use affect streamflow and temperature during low-flow conditions.

5.0 CONCLUSION

This study investigates water management in a decentralized management regime in coastal California. First, a GIS is used to determine existing storage locations in the watershed and the EPV method was developed and used to estimate the maximum amount of water that is stored in each pond. This study also investigates spatial and temporal patterns of intermittency and potential impairment due to filling private reservoirs, particularly during low-flow conditions. Finally, this study investigates patterns of instream water temperatures and climatic conditions and how that might affect the survivability of salmonids in the watershed.

Understanding demands of human water use coupled with aquatic needs is key to creating effective watershed management policies. However, decentralized watershed management can be complicated by the multitude of stakeholders and policies that apply to different types of water rights unevenly. This study demonstrated that EPV could be a useful approach for estimating annual water stored in private reservoirs in areas that don’t have water rights reporting or lack accuracy.

This study demonstrates that late season intermittency should be considered, in addition to early season intermittency, as a limiting factor for salmonids particularly following periods of drought. The end of intermittency was found to persist later into the water year when followed by a dry-type year. In addition to ending later, intermittency appears to be occurring more frequently and for longer periods than it did historically. Longer durations of intermittency were observed following cumulative years of dry-type conditions. The cumulative effects of dry-type conditions could affect rearing salmonids that need to rear for 1-2 years or more in the stream (NMFS, 2008; NOAA, 2012).

The use of private reservoirs to maintain flow during low-flow conditions is being promoted in the watershed (SRCD, 2015); this study highlights that the timing private reservoirs are refilled is likely important for managers to consider in the Maacama watershed. This study shows that private reservoirs have the potential to extend intermittency which could affect salmonid life stages and coincide with migration and spawning, particularly during periods of drought. Additional analysis is needed to determine whether flows are adequate during salmonid migration and spawning periods following drought. In addition, more research is needed to determine if adequate habitat exists to maintain ecological processes during low-flow conditions as the rainy season resumes and if early season withdrawals affect rearing salmonid survival.

Current management practices either lack restrictions on the timing of refilling reservoirs or are not effective during periods of drought, which could be potentially exacerbated by climate change. Adaptive management practices that can adapt to drought conditions for all water users, which includes altering the timing of when reservoirs are refilled, would be beneficial. In addition, adaptive management would be more effective if all water use was accurately recorded and reported to the state.

Finally, water temperatures were above desired salmonid growth thresholds for every year that was included in this study. In addition, coho were unlikely to have survived the high maximum temperatures.

The timing of diversions and the ability of policy to implement, monitor and adaptively manage water withdrawals for private reservoirs over the entire watershed will likely determine the success of these approaches. Monitoring and management policies that can adapt and be implemented in response to drought conditions will likely be more successful at maintaining instream flow for ecological health.

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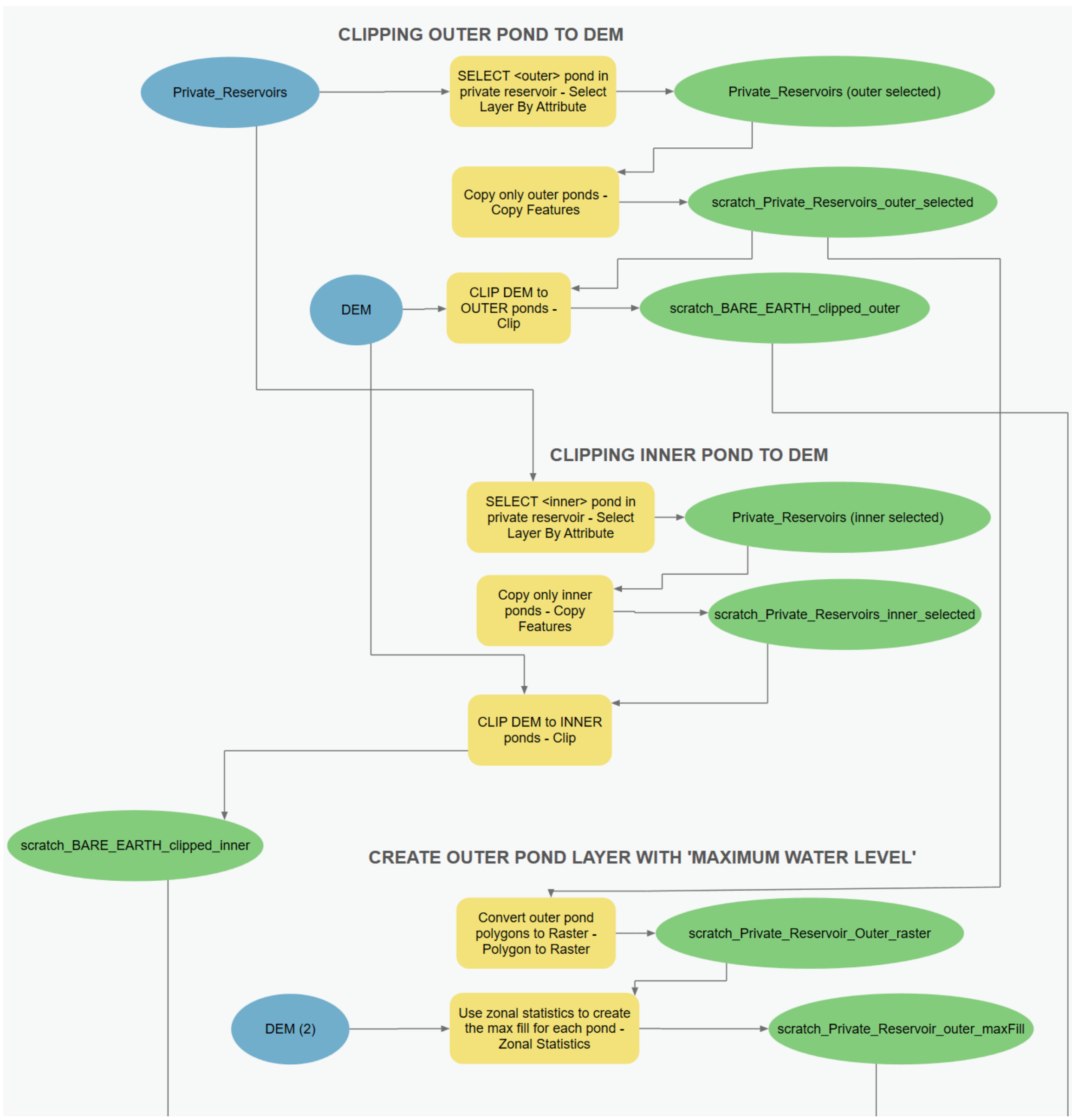
APPENDICES

APPENDIX 1: EFFECTIVE POND VOLUME (EPV) MODEL



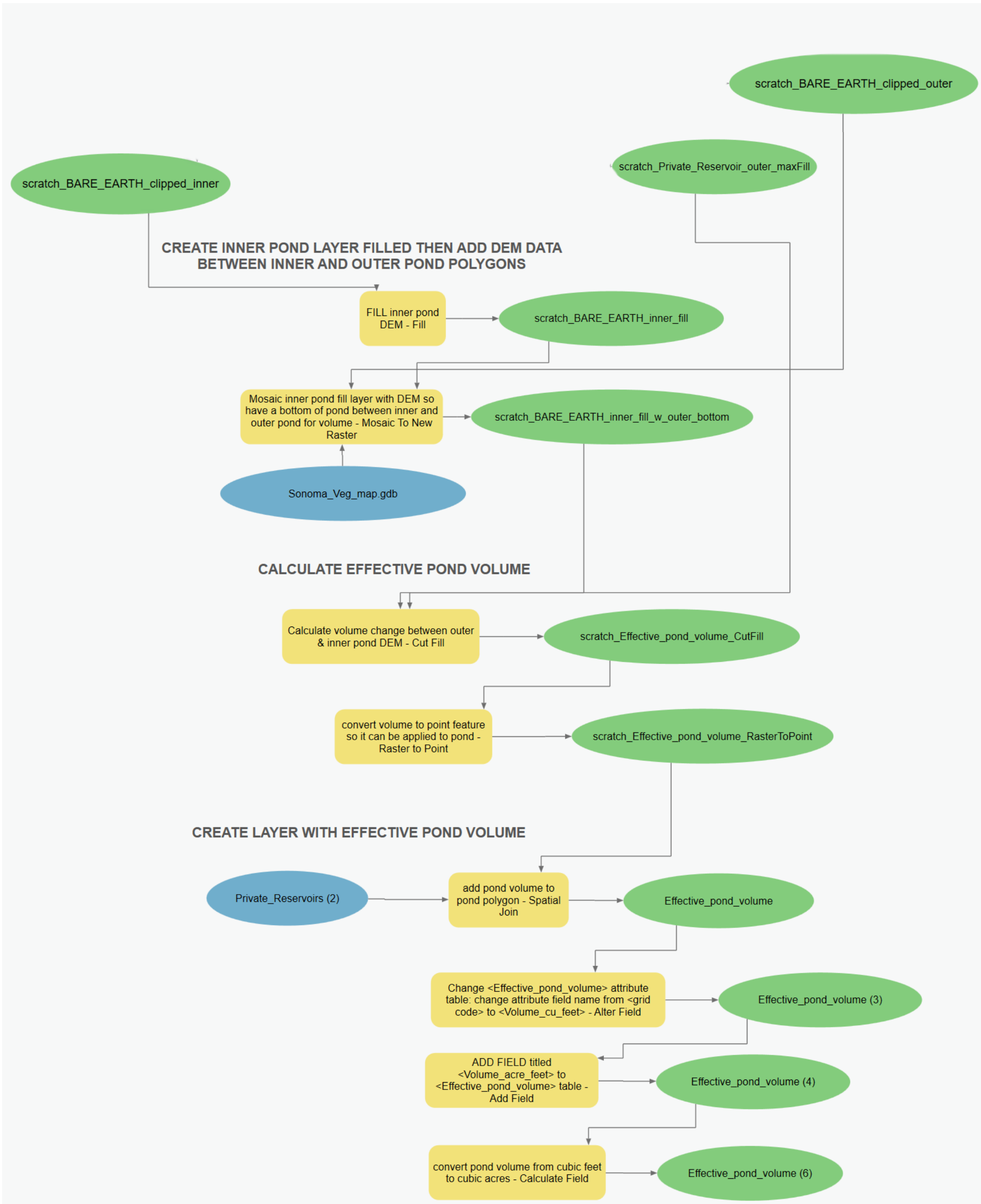


Figure 22: Effective Pond Volume (EPV) Model Diagram

The steps taken in the model are as follows

1. The clip tool was used with the Outer Ponds as the Input and the DEM as the Clip Feature. The Output was a DEM for each Outer Pond.

2. Next the Clip Tool was used with the Inner Ponds as the Input and the DEM as the Clip Feature. The Output was a DEM for each Inner Pond.

3. Next the Fill Tool was used on the Outer Pond layer to create a Maximum Water Level for each Outer Pond.

4. Next the Fill Tool was used on the Inner Pond DEM and then added the DEM data for the portion of the pond between the Inner and Outer ponds.

5. Finally, to create the Effective Pond Volume the CutFill do you spacetool was used to calculate the Volume between each Inner and Outer pond and added that volume to a new layer named <Effective_pond_volume> then converted the value EPV from cubic feet into acre-feet for easy comparison to eWRIMS data.