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Eco-geomorphic flows: Modification of wet-season dam operations to support downstream salmonid habitat

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Eco-geomorphic flows: Modification of wet-season dam operations to support downstream salmonid habitat

By

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Eco-geomorphic flows: Modification of wet-season dam operations to support downstream salmonid habitat

Abstract

Most large rivers in California and much of the world have dams impeding their flow. These dams provide important benefits to society but have negatively affected river ecosystems and the aquatic organisms that live in them by disrupting the natural patterns of movement of water and sediment. To mitigate some of these harmful effects, river managers often turn to physical habitat restoration and environmental flow manipulation. All dams have designed reservoir release operations and many rivers downstream of them have habitat restoration projects, but rarely do managers design these to work together to yield the best combined outcome. This study investigates the potential to modify flood risk management releases to better support downstream habitat enhancement projects designed to provide endangered salmonid rearing habitat. This goal is accomplished using an hourly reservoir operations model, a one-dimensional sediment transport model, and a two-dimensional steady flow model combined with habitat suitability analysis. Results show that lower magnitude, longer duration flood releases have the potential to increase provision of rearing habitat during fry and juvenile rearing periods while also reducing operations and maintenance costs for the constructed restoration projects and providing ancillary benefits to water supply. Furthermore, because of the changed river corridor and sediment regime, the results indicate that pre-dam flows would not be ideal for meeting current management objectives of providing rearing habitat below the dam, and that flows reconciled to the current physical, ecological, and operational contexts would be more appropriate.

1.Introduction

Most large rivers in the western United States and much of the world have dams altering their flow regime (Zarfl et al 2014, Cooper et al 2017). These dams provide important benefits to society, including water supply storage, flood risk management, hydropower and recreation (Ho et al 2017, Llamosas and Sovacool 2021). Most of these large dams were built during the early to mid-20th century prior to modern environmental laws (Hanak 2011, Song et al 2021). As a result, they have often had severe negative consequences to river ecosystems, interrupting longitudinal connectivity, forcing relocation of communities, and disrupting the natural patterns of movement of water, sediment, and biota throughout river basins (Cooper et al 2017, Ho et al 2017). This disruption of water and sediment harms downstream ecosystems beyond the river discontinuity from the dams themselves (Schmidt and Wilcock 2008).

Changing the timing of water flow through rivers is a necessary and unavoidable side effect of storing water in reservoirs - they facilitate the movement of water through time from when it is available to when it is needed. Natural flow regimes have an intrinsic seasonal and interannual variability, especially in rivers in the western U.S. (Poff et al 1997, Dettinger et al 2011). Dams reduce this natural variability by creating a physical barrier across the river to store water and regulate outflow. In the western U.S., flood risk management and water supply are major purposes of large dams, so the management objective is usually to capture large floods in winter (reducing flood damages downstream) and release the water for agricultural and urban uses in summer (Ho et al 2017). When river flow regimes are highly altered for meeting human water management objectives, they often lose their ability to support natural processes and ecosystems (Zarfl et al 2014).

There are well-known interactions between dams and sediment supply and transport in rivers (Kondolf et al 2014, Randle et al 2021). Dams interrupt the transport of coarse sediment, which frequently leads to a state of sediment deficit and corresponding incision, floodplain abandonment, and general simplification of the river corridor downstream (Schmidt and Wilcock 2008, Walker et al 2020,

Randle et al 2021). Alternatively, aggradation can occur instead if rivers have significant tributary sediment input downstream of the dam, especially if there is a significant water diversion or the changed hydrograph from dam releases lacks sufficient stream power to transport that sediment (Schmidt and Wilcock 2008). As sediment transport is a non-linear process with sensitive dependence on numerous natural and anthropogenic factors, this channel response is difficult to predict and model accurately (Ancey 2020).

The recognition of the harmful effects of flow and sediment regime modification by dams has driven a push to re-operate reservoirs to provide environmental flows that mitigate some of these harmful effects while staying within operational constraints (Acreman and Dunbar 2004, Richter and Thomas 2007, Poff et al 2010). Incrementally restoring components of the natural flow regime can benefit downstream species and improve downstream ecosystem function (Poff et al 1997, Acreman et al 2014, Sandoval-Solis and McKinney 2014, Lane et al 2015).

Rising in parallel to environmental flows, river restoration is an emergent field that has typically focused on adjusting channel form and active channel processes to improve habitat for freshwater and riparian species (Papangelakis and MacVicar 2020, Pasternack 2020, Ciotti et al 2021). Its application varies by river and habitat type, but frequently a desired goal is to create more off-channel or floodplain habitat (Bernhardt and Palmer 2011). There is little agreement regarding the methodologies used to evaluate and design river restoration projects, but there is a general consensus that some form of physical habitat restoration could provide a significant benefit in most degraded river ecosystems (Simon et al 2007, Wohl et al 2015b). The difficulty of predicting channel response following damming and other channel perturbations frequently presents itself in the context of process-based river restoration, which aims to harness these river processes to improve habitat for riverine species (Ciotti et al 2021). There have been many examples of river restoration projects that have failed to achieve their goals because river processes like sediment transport were not adequately considered (e.g. Kondolf 1998), and many others

that have been successful when these processes were taken into account (e.g. Staentzel et al 2020, Ciotti et al 2021).

Restoration projects are frequently downstream of dams, and so the flow regime is a critical component of these projects, especially as it provides connectivity to habitats adjacent to the main river channel (Munsch et al 2020). Flow has been found to be a dominant determinant of habitat for endangered fish, while channel form can dominate geomorphic processes (Lane et al 2018). Because dams control flow in so many rivers, they will be an essential tool in restoring habitat for sensitive species, particularly when used in conjunction with process-based restoration projects (Beechie et al 2012, Whipple and Viers 2019).

In projects downstream of large federal dams, the flood risk management component of dam releases is usually considered unchangeable because it based on water control manuals that are rarely updated, i.e. it is an existing condition to incorporate into the project design (Fennell et al 2016, Patterson and Doyle 2018). Dam flood releases are by definition controlled by river managers and so represent a valuable opportunity in restoring river functionality for aquatic species (Whipple and Viers 2019). However, there are operational limits for implementation of environmental dam releases that need to be considered, including a limited availability of water for many different competing purposes, and these will only be exacerbated by continued climate change (Swain et al 2018, Delaney et al 2020).

The goal of this study was to investigate the impact of modifying dam flood risk management operations to obtain more functional downstream river processes in support of river restoration sites designed to provide endangered salmonid rearing habitat. The main hypothesis is that the provision of salmonid rearing habitat within restoration projects and throughout the river corridor can be improved by designing flood releases that control sediment transport processes at tributary junctions while staying within operational limits. The specific objectives of this study include: (1) generate and evaluate alternative reservoir flood release strategies and their effects on water supply and flood risk management, (2) characterize relative effects on sediment transport and other hydraulic

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variables with these release strategies, (3) estimate habitat performance in restoration sites and throughout the river corridor for sensitive life-stages of species of concern, and (4) compare performance of the different release strategies relative to watershed management goals. Warm Springs Dam (WSD) located on Dry Creek in Sonoma County, CA, where dam flood risk management operations in recent years have negatively impacted restoration sites, is used as a case study.

2.Literature Review

2.1.Alteration of water and sediment regimes

The operating rules that guide reservoir releases are established by state, federal, and local agencies that operate hydro-structures for a given set of operational objectives (e.g. water supply, flood management, hydropower, sediment control, etc.). For large federal projects, flood control rules are set in water control plans written by the U.S. Army Corps of Engineers (USACE 2017). Water control plans typically separate the reservoir storage into the conservation pool, where water is managed for water supply, and the flood pool, where water is managed to reduce the risk of downstream flood damages. One of the guiding flood risk management principles is that when the reservoir storage is in the flood pool, water should be evacuated as quickly as possible without exceeding the safe rate of release, which is typically taken to be the downstream channel capacity (USACE 2017).

The way reservoir releases are currently managed in most dammed rivers has shortcomings from both societal and environmental perspectives (Castelletti et al 2008). These rules are rarely updated, as they require lengthy administrative processes and strong justification to change (Patterson and Doyle 2018). Policy inertia in the United States makes it so that reservoir release practices are unlikely to change in the absence of a serious conflict or poor management outcome (Giuliani et al 2014). The strictness of these rules can lead to detrimental consequences.

An example of this occurred during Water Year 2015 near the case study area, when a large storm in December 2014 filled Lake Mendocino near Ukiah, CA above its flood pool capacity, which led the dam operators to release large amounts of water to evacuate the flood pool according to the water control plan for the project. However, because there was little precipitation for the remainder of that winter, the water control plan's mandated releases resulted in a near-critical water shortage during the following irrigation season, which could have been prevented with more flexible operating rules (Jasperse et al 2017, Delaney et al 2020). Situations like this have driven the push for the revision of dam operation rules from a human use perspective.

From the environmental perspective, the flow effects of dams can vary widely based on dam operational rules. The often-radical changes to the timing, duration, and magnitude of low and high flow periods downstream of dams can significantly affect aquatic ecosystems and are captured in metrics like those in the Indicators of Hydrologic Alteration (IHA) methodology (Richter et al 1996). Various other statistics to characterize the degree of hydrologic change have been developed since then, including the concepts of ecodefecit and ecosurplus, which attempt to simplify the statistical analysis and facilitate development of environmental flow regimes (Gao et al 2009). Other more recent developments include the use of statistical modeling and landscape-scale datasets to characterize the degree of flow alteration (e.g. Carlisle et al 2010, Arthington et al 2018).

Another major component of flow through river systems is sediment, which can vary in size from fine particles (i.e. clay, silt, and sand) to coarse particles (i.e. gravel, cobble and boulders). Sediment not only moves through rivers, it also makes up the bed and banks of channels and provides habitat for aquatic organisms. Many species depend on particular size distributions of riverbed materials for various life stages (Groot 1991, Kondolf and Wolman 1993, Montgomery et al 1996, Groves and Chandler 1999, Wohl et al 2015a). Typically, dams trap almost all coarse sediment moving down the rivers from their watersheds, thus reducing reservoir storage capacity and depriving downstream river reaches of material essential for channel formation and aquatic habitats (Kondolf et al 2014).

River systems are constantly in flux, and as flows of water and sediment change, the channel changes with them. The Lane balance (Figure 1) is a classical conceptualization of how rivers maintain a dynamic equilibrium with varying stream power of water (steep or shallow slope) and sediment (coarse or fine and total load), and whether that leads to aggradation or degradation (Lane 1955, Dust and Wohl 2012). Following dam construction, rivers go through a period of channel adjustment as the sediment balance changes (Phillips et al 2005). Most regulated rivers in the western U.S. are currently in sediment deficit as a result of this process and consequently going through a process of bed degradation and channel simplification (Schmidt and Wilcock 2008). However, the direction and magnitude of channel change following dam construction can often be difficult to predict because of other processes that affect sediment transport occurring in watersheds such as logging, grazing, mining, water diversions and other land uses (Nelson et al 2013). Channel change prediction is further confounded by coarse sediment transport being a highly nonlinear and difficult-to-measure process (Wohl et al 2015a, Ancey 2020). Despite these issues, a thorough understanding of sediment transport is important for managing habitat downstream of dams.
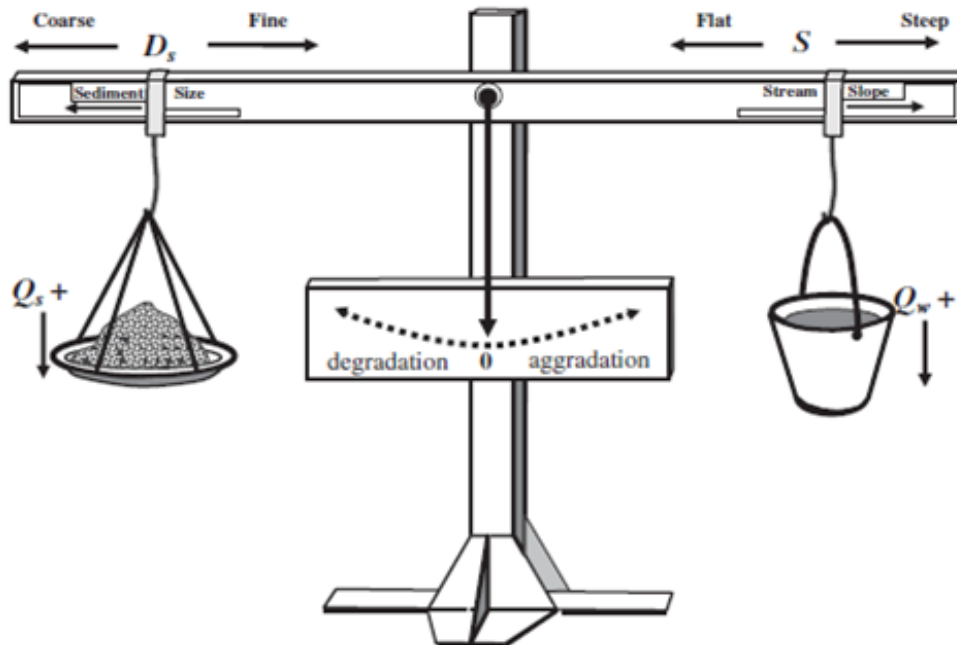
Figure 1. Depiction of Lane's relation as a balance. Reprinted from Dust and Wohl 2012.

The harmful effects of dams are well established, and a movement has since emerged to try to mitigate those effects (Ligon et al 1995, Kondolf et al 2014). Dam removal is an occasionally implemented and often very successful strategy. Dam removal quickly reestablishes ecosystem connectivity, sediment continuity and other ecosystem processes and functions (Foley et al 2017). On the Elwha River in Washington, this removal quickly redistributed stored sediments through the river and its delta, and had significant benefits to aquatic ecosystems (Warrick et al 2015). The Elwha was a unique case, because there was only a short distance from the ocean for sediment to impact, whereas many large dams feed >100-km lowland river segments that are more susceptible to sediment impacts (Major et al 2017). The complete removal of dams is often impractical because of the important human uses they serve and concerns over the loss of mitigation a dam provides against catchment-scale anthropogenic impacts hurting lowland communities (Shields 2009, Vigars 2016). Removal of some dams will continue as their useful economic and operational life expires, and the tradeoffs between habitat gained vs. water supply and flood control capacity lost will need to be evaluated (Null et al 2014).

2.2. Toward Environmental Flows

Short of complete dam removal, environmental flows, defined here as flow releases from dams specifically intended to benefit downstream ecosystems, can be a useful reconciliation strategy to improve ecosystem health while still maintaining the benefits of storing water behind dams. The goal with many environmental flow regimes is to match the natural variability in the system using reservoir releases during ecologically important time periods (Poff et al 1997, Tharme 2003, Poff et al 2017). The methods used to arrive at flow guidelines for any particular river will vary based on national and state laws/policies, hydrologic context, specific management goals and resources available, but can broadly be categorized as hydrologic, habitat-simulation based, or holistic (Tharme 2003, Acreman and Dunbar 2004).

Hydrologic methods focus on the analysis of existing or historical streamflow data to arrive at a recommendation of instream flow requirements. The recommendation can be a percentage of the natural flow regime at a given time of year, or focus on ecologically relevant flow timing, magnitude and duration for species of concern in the study system (Richter et al 1996, King and Louw 1998, Tharme 2003, King and Tharme 2008, Poff et al 2017, Patterson et al 2020). Habitat simulation methods use known relationships between flow, hydraulic variables like depth and velocity, and species utilization to prescribe flows that maximize habitat suitability (Bovee 1982, Bovee et al 1998, Acreman and Dunbar 2004, Gregory et al 2018). Holistic methods generally combine scientific expertise with local stakeholder knowledge to integrate hydrologic or habitat simulation methods with local management goals and constraints (Tharme 2003, Poff et al 2017). Examples of these include the Downstream Response to Imposed Flows Transformation (DRIFT) framework and the Instream Flow Incremental Methodology (IFIM), both of which attempt to incorporate all relevant biotic and abiotic factors, including legislative and geographic context, to come up with a comprehensive flow regime (Bovee 1982, Bovee et al 1998, King et al 2003, Poff et al 2017).

These methods all have in common that they attempt, directly or indirectly, to relate amounts of water at varying times of the year to habitat or ecosystem quality. However, sediment supply is often not explicitly considered in these methods of establishing environmental flow regimes, except in the case of holistic frameworks like DRIFT and IFIM (King et al 2003, Bovee et al 1998). If the goal is to provide more habitat for sensitive species and ecosystems, approximating a pre-dam flow regime (or its functions) will not be effective if no consideration is given to the sediment supply available for channel forming and maintaining processes (Wohl et al 2015a). For example, a spawning freshet meant to clear fine sediments from spawning gravel for salmon is meaningless if no spawning gravels are present because they have been eroded from below the dam and no additional supply is coming into that reach. Along with the natural flow regime, there needs to be consideration of the sediment regime (Richter et al 2006, Wohl et al 2015a).

Wohl et al 2015a makes the case that it is essential for river managers to consider the sediment supply in a river reach or basin when prescribing flow regimes. While the natural or pre-modern era flows of water can be approximated with controlled dam releases, this is almost never the case for flows of sediment. Beyond the effects of the dam in interrupting coarse sediment continuity through the watershed, the myriad effects of logging, grazing, mining, urbanization, gravel extraction and other activities can change the timing, volume and size of sediment entering the system (Magdaleno et al 2018). Characterizing these inputs and outputs in a sediment budget and considering the river's ability to transport that sediment can allow river managers to identify flows that can maintain the river channel in dynamic equilibrium with the sediment supply (Jurotich et al 2021). Because habitat structure is fundamental to aquatic species, it is essential to consider water and sediment together in post-disturbance reconciliation of river ecosystems (Wohl et al 2015a).

The functional flows methodology operationalizes this framework by focusing on both geomorphology and ecology (Escobar-Arias and Pasternack 2010, Yarnell et al 2015). Escobar-Arias and Pasternack 2010 specifically define a functional flow "as a discharge that interacts with river bed morphology through hydraulic processes providing a shear stress value that serves an ecological function." This is related to the habitat simulation methods mentioned above, with a strong emphasis on mechanistically linking the hydrology, geomorphology and ecology of a river. It describes the environmental flow in the context of a river's sediment supply and ecological functions desired to be achieved by the flows, following Wohl et al 2015a. The functional flows framework is developed in the context of highly modified rivers in a Mediterranean montane climate in the western U.S. but could be modified and applied to other systems. The Yarnell et al 2015 and 2020 framework conceptualizes the river's flow as five components and discusses each in the context of its ecosystem functions (physical, biogeochemical and biological) (Table 1).

Table 1. Functional flows and their ecosystem effects, from Yarnell et al 2015 and Yarnell et al 2020.

Flow Component	Physical Ecosystem Function	Supported Ecosystem Function
Wet-Season Initiation	Flush accumulated hillslope sediments, flush fines from spawning gravels	Seasonal nutrient cycling, migration cues, reactivate connectivity with hyporheic zone
Peak	Channel restructuring and rejuvenation by redistributing sediment and large wood	Reset riparian vegetation, allow species to access floodplains
Spring Recession	Gradual sediment deposition in shallower habitats	Reproduction and migration cues, riparian seedling germination, juvenile fish rearing
Dry-Season Baseflow	Prevent silt accumulation (can occur if baseflow is artificially high)	Restrict connectivity and promote ecological niche partitioning
Wet-Season Baseflow	Increase longitudinal connectivity	Support migration, spawning, and residency of aquatic organisms, and growth of riparian habitat
Interannual Variability	Support long-term diversity of habitat conditions in adapted ecosystems	Regulate river food webs and reset successional patterns

For example, the first is wet-season initiation flows. These flows transition the system from dry season to wet season, and in some systems initiate seasonal nutrient cycling and cue upstream migration for anadromous fish. They also flush accumulated hillslope sediments downstream and flush organic material and fines from spawning gravels. The other components of the flow regime discussed (Table 1) are peak magnitude flows, spring recession flows, dry-season low flows and interannual variability (Yarnell et al 2015). Subsequent work has also added a wet-season baseflow, which contributes to important elements of migratory fish habitat (Yarnell et al 2020). While this framework has promise for improving habitat outcomes, it has yet to be implemented on a large scale.

2.3. Operational Flexibility

Instream flows that consider a combination of geomorphic and ecological aspects of flow regimes are an important step forward, but factoring in the human uses of water supply and flood risk management complicates their implementation. Implementing an environmental flow regime without taking away from water supply and flood risk management capacity requires that systems have considerable flexibility in how they are managed, as well as a relatively comprehensive understanding of downstream habitat needs to assess effects of different flow regimes on downstream ecosystems (Adams et al 2017). In many rivers of the western U.S., human demands have often conflicted with environmental uses because of their timing. Irrigated agriculture requires water during the growing season typically during the dry months of the year, while many ecosystem demands are in the wet months, especially in rainfall dominated systems where large floods are stored and peak flows reduced in magnitude (Niu et al 2019). As rivers are increasingly managed for environmental flows downstream, it is important to consider the impacts to agricultural communities and other water users (Grantham et al 2014).

Beyond this conflict, climate change is predicted to increase variability in amount and timing of precipitation, which will lead to increase in extreme droughts and floods in the western U.S. and further complicate questions around re-operating dams to accommodate river restoration and environmental flows (Capon et al 2018, Swain et al 2018, Delaney et al 2020). While the environmental effects of dams are generally net negative, there are some important positive effects that should be considered, especially in the context of climate change. They can provide reservoirs of cold water that can be used to sustain fish populations imperiled by increasing water temperatures elsewhere in the watershed and prevent the migration of invasive species (Beatty et al 2017). For the foreseeable future, reservoirs will continue to be managed for human needs, so increased flexibility is necessary if trying to alter reservoir releases to meet environmental needs while adapting to the realities of climate change. Numerous strategies have been proposed for modernizing reservoir operations, but generally they include using modern flood forecasting tools and/or incorporating other uses like environmental flows or conjunctive use (Nayak et al 2018).

Current rules dictating flow releases for most USACE-managed reservoirs (i.e. water control plans) were developed at the time of reservoir construction (Patterson and Doyle 2018). Some rules have changed to reflect shifting demands and minimum instream flow legal requirements, but flood management rules dictating winter baseflow and peak flow magnitudes are largely the same (Fennell et al 2016). Forecast Informed Reservoir Operations (FIRO) is one method suggested to increase flexibility in wet-season reservoir operations and conserve more water for later use (Ralph et al 2014). This method uses ensemble streamflow forecasts based on uncertainties in future atmospheric conditions to predict potential future reservoir inflows for different scenarios (Delaney et al 2020). Individual ensemble traces are then evaluated to see if they will cross a flood control threshold. The combined ensemble risk is then used to determine actual releases (Delaney et al 2020). In the case study presented in Delaney et al 2020, FIRO yielded a 33% increase in median storage over existing operations at the end of the flood management season. This operations strategy results in more water with which to make release decisions while meeting all flood constraints. However, it still assumes that once minimum instream flows are met, there are no further environmental needs. FIRO could be combined with a functional flows framework-type approach to use the extra conserved water for eco-geomorphic purposes that support and enhance downstream ecosystem health.

Environmental hedging is another approach intended to add management flexibility by maximizing environmental benefits while still meeting human demand needs (Adams et al 2017). Adams et al 2017 present an optimization model intended to decide reservoir releases during different parts of the fall-run Chinook salmon life cycle to maximize fish population in the long term. In many water management system optimization models, minimum environmental flows are used as a constraint, but here water supply for downstream users is used as a constraint instead. Once that demand is met, water can be used for fish lifecycle demands. The hedging part of the approach comes from the idea that small shortages are acceptable or even desired if they minimize the risk of large shortages in the future. The approach is particularly useful for anadromous fishes that have several age classes, each of which is

necessary to ensure total population viability. It is also useful in elucidating the tradeoffs associated with providing water for the environment during different times of the year (Adams et al 2017). The approach taken in Adams et al 2017 and other similar studies that use a formal optimization process to set environmental releases show promise for moving away from a rule-based to an outcome-based approach to environmental flow management (Horne et al 2018). Transitioning to an outcome-based mindset is particularly important given the magnitude of changes that have occurred to river ecosystems since the early 20th century and the pressing need for action given the high extinction risk of many sensitive aquatic species (Zarfl et al 2015, Katz et al 2013).

2.4. Managing Altered Systems

Following damming, rivers can become novel ecosystems, in that they take on a different ecosystem trajectory directly as a result of human intervention (Morse et al 2014). In California, the various types of stream alteration were categorized by Guitron 2020, and they include impacts from various land uses such as urban, agriculture, logging and reservoirs (Guitron 2020, Figure 2). The novel ecosystems that occur from damming highlight the importance of an outcome-based approach (Horne et al 2018). Because dammed rivers have all their coarse sediment intercepted and often a lot of their flow diverted, they effectively become different rivers (Moyle 2013). This calls into question the effectiveness of using pre-dam flows and habitat conditions as target instream flows, and supports the idea that modern management of water should be reconciled with what is possible in the system in question (Zedler et al 2012). Several examples of this kind of management exist in the rivers of California and elsewhere in the western U.S.
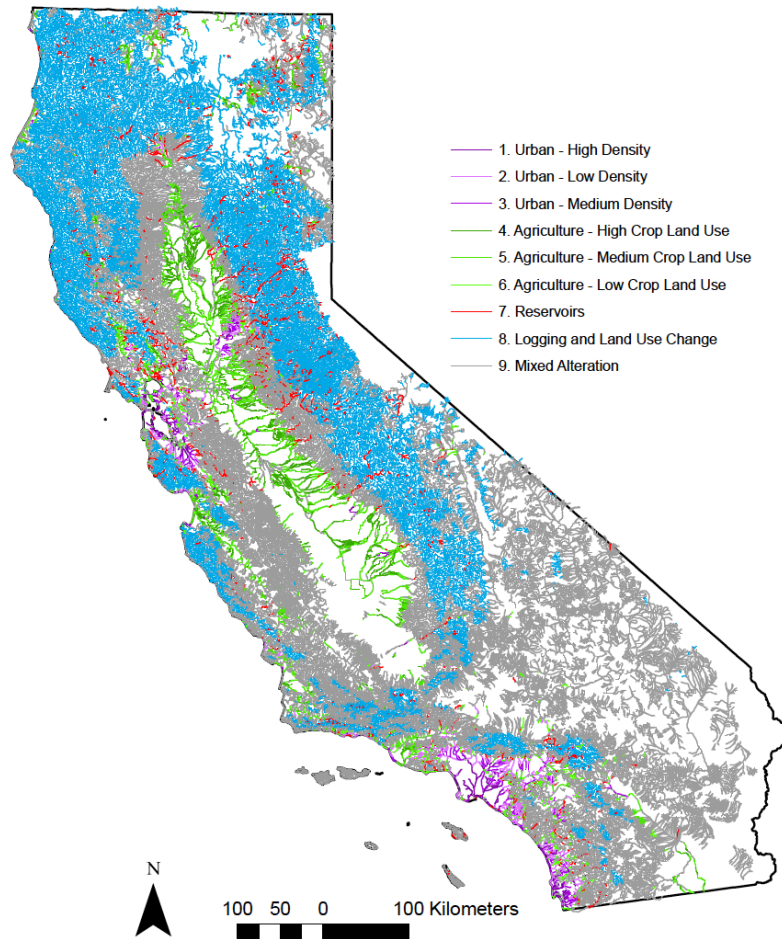
Figure 2. Spatial distribution of altered stream types in California, reprinted from Guitron 2020.

After the closure of a system of dams on the Trinity River in northern California, major changes were observed in the downstream geomorphology from diversion of large volumes of water and interception of coarse sediment (Viparelli et al 2011). In the immediate aftermath, the river lost its ability to transport sediment supplied to it by tributaries and the channel network became greatly simplified due to vegetation encroachment. To remedy this, managers have supplemented the gravel supply and implemented an engineered flood flow regime to restart geomorphic processes to ensure suitable anadromous fish spawning conditions(Viparelli et al 2011). This management approach occurred concurrently with physical habitat restoration projects and had the goal of moving towards a reconciled ecosystem that is a smaller version of its former self (Ock et al 2015).

The middle reaches of the Colorado River are another place where reservoir releases have been used to meet specific geomorphic and ecological goals (Mueller et al 2018). The post-dam environment is a heavily altered system with large reservoirs that can hold much more water than the mean annual flow. These reservoirs along with water diversions from various users have resulted in a novel ecosystem that presents unique management challenges. Reaches between reservoirs have a tributary input of sediment available for management goals. From the mid-1990s to the mid-2000s, river managers implemented a series of test floods with varying co-equal goals related to flow/sediment management, riparian zone vegetation management and maintenance of open sandbars for recreation (Stevens et al 2001, Mueller et al 2018). These test floods showed that it is possible to have sediment deposition even in a system where dams intercept much of the sediment if flows are timed to coincide with tributary inputs of sediment, but that it is highly sensitive to site-specific characteristics of topography and vegetation dynamics (Mueller et al 2018). The results from these floods resulted in changed management protocols that focus on the geomorphic work that particular flood flows can do (Mueller et al 2018).

Another challenge in implementing a functional flows-like approach is the individuality of each river. In some rivers, small, controlled floods are sufficient to do the work required for maintenance of habitat structure suitable to the post-disturbance river ecosystem (Rose et al 2018), but this will vary with sediment supply and water diversion from the system.

The variability in management outcomes is illustrated by two different cases in Arizona. The Bill Williams River was dammed in 1968 and managed to eliminate natural floods from the system. The altered flow and sediment regime promoted invasive species and prevented the establishment of native vegetation (Glenn et al 2016). River managers collaborated to release experimental floods and saw success in reversing these changes, albeit at a smaller scale (Glenn et al 2016). The successes from this effort were attempted to be transferred to the Minute 319 order, which was a binational agreement in 2012 that called for environmental flows in the Colorado River delta (Pitt and Kendy 2017). This was a good example of cooperation between a wide variety of stakeholders operating in a constrained environment,

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but while it had some temporary benefits to the riparian ecosystem, it resulted in recruitment of invasive vegetation and did not provide enough water to sustain important ecological processes over the long term (Glenn et al 2016). The operational flexibility that allowed for the long-term shift in operations of the reservoir on the Bill Williams River was the key difference between these two systems.

Flexibility is needed to manage rivers into the future for both human and environmental benefits. The umbrella of methods known as adaptive management has flexibility as one of its central goals. It provides a formalized system of evaluating outcomes resulting from management actions and adjusting them to achieve predetermined management goals (Holling 1978). It is well-suited to environmental flow programs, and when the process is well-documented, it can inform future decisions in other systems (Webb et al 2018).

The push to incorporate environmental flows in operation of dams, while promising, needs to be paired with clear and measurable goals, physical habitat restoration and an understanding of the remaining sediment supply in the river system to be fully successful (Pasternack 2008, Whipple and Viers 2019). While there has been a lot of study related to the design of restoration sites, there has not been as much work related to explicitly developing flood management operations rules to maintain these restoration sites in the long term (Larrieu and Pasternack 2021). This study explores the concept of developing an eco-geomorphic flow regime (i.e. one with both ecological and geomorphic functions in mind) to support endangered species habitat and emphasizes the importance of contextualizing this flow regime for each river's sediment supply and geomorphological setting. The Trinity River example, while illustrative of a system that has used both physical habitat restoration and environmental flows to support salmonid habitat in an adaptive management framework, has not explicitly combined both actions to maximize the success of each. This study is intended to provide an example of adjusting flow release-supported river process management to support physical habitat restoration sites, a need identified by Whipple and Viers (2019), while taking into account impacts on current operations. The methodology

used here is broadly applicable to river corridors downstream of dams where managers are trying to meet specific ecological goals.

3.Study Area

Dry Creek is the largest tributary in volume to the Russian River in coastal Northern California, USA (Figure 3). The region experiences a Mediterranean climate and corresponding flow regime with potentially large, rain-dominated winter floods and a long, dry summer season. The watershed is approximately 559 km^2 (216 mi^2) and has a mean annual precipitation of 1318 mm (51.7 in) (USGS streamstats). WSD and Lake Sonoma (LS), which are managed by USACE during flood risk management operations and Sonoma Water during water supply operations, were built in 1983. Flow modification from WSD has greatly reduced the magnitude of winter floods and increased flows during summer months, which although it has provided a stable source of water supply for urban and agricultural users, has also led to significant changes in the ecology and geomorphology of the river corridor.

WSD captures runoff and prevents transport of coarse sediment from approximately half of the Dry Creek watershed area. The upper watershed is mountainous and forested, while the valley floor is relatively flat and extensively cultivated for wine grapes. Pre-dam peak flows were as high as 1,132 cubic meters per second (m^3/s, equivalent to 40,000 cubic feet per second [cfs]), while post dam flows are controlled releases that rarely go above 142 m^3/s (5,000 cfs). Several tributaries enter Dry Creek below the dam and contribute water and sediment. The largest of these tributaries are Pena Creek, approximately 4 miles below the dam, and Mill Creek, close to the confluence with the Russian River. Dry Creek is home to several species of threatened and endangered salmonids that have experienced a precipitous decline in population over the past 100-150 years. There were around 20,000 spawning Coho salmon adults that returned to the Russian River watershed each year pre-1900, but this number had dwindled to below 10 in the early 2000s (NMFS 2012).
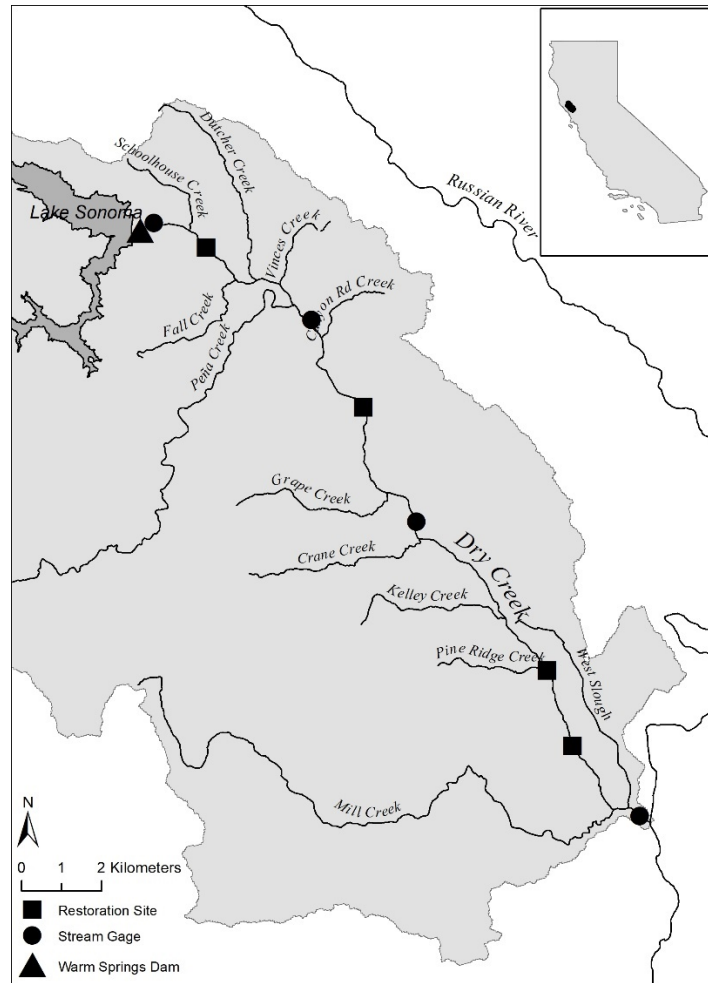
Figure 3. Study overview map with regional location in CA shown.

The region's geomorphic history was well-studied and documented as part of the dam construction process. Logging and land-use changes first led to aggradation, and then base-level lowering as a result of gravel mining led to several cycles of incision and narrowing of the river corridor (Harvey and Schumm 1987). The end result of this is an incised river corridor with bottom elevations approximately 6 m below the vineyard terrace level. Since construction of the dam, a mature, even-aged riparian canopy, several grade control sills, and naturally occurring bedrock have severely limited channel migration, and the dam prevents large floods that would otherwise provide a periodic reset of successional patterns and geomorphic form. Prior to damming, the creek would go dry in the summer months, and experience large floods in the winter months. These large floods performed important ecological

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functions such as supplying and mobilizing gravel, and providing conditions for willow, cottonwood and alder seedlings to establish (McBride and Strahan 1984, Magdaleno et al 2018).

During flood risk management season, WSD holds back water during storm events to minimize downstream flooding. When storm-driven floods subside, WSD releases this water to make more room in the flood pool. The water control plan has flood control schedules that set various constraints, such as ramping rates and maximum flows, on the releases (Figure 4). Important in the context of this study, the rule curve for WSD does not include a seasonal component - the top of conservation pool is consistently set to be 302 million m^3 (245,000 acre-feet) and does not increase in the spring, summer, and fall months (USACE 1984). Dam releases are further regulated by the National Marine Fisheries Service (NMFS) through their administration of Section 7 of the Endangered Species Act. NMFS' most recent biological opinion found that the continued operation of WSD had the potential to jeopardize continued survival of the Russian River population of Coho salmon, and recommended as a Reasonable and Prudent Alternative that Sonoma Water and USACE establish six miles (9.7 km) of habitat enhancement projects to provide salmonid rearing habitat in Dry Creek below WSD (NMFS 2008).
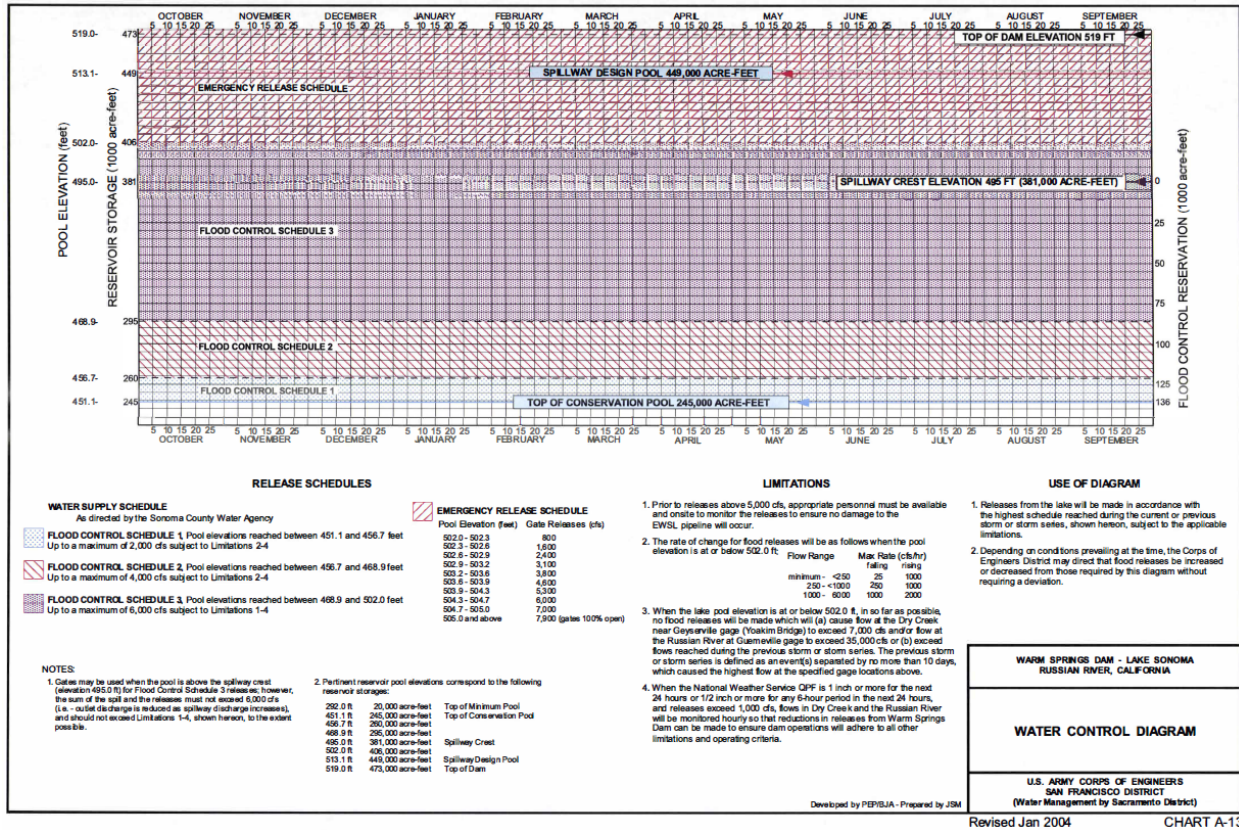
Figure 4. Water control diagram from WSD water control plan.

The post-dam channel form and flow regime were such that summer velocities were too high for juvenile coho salmon and steelhead, so the focus of the projects was on low velocity, relatively shallow pools with sufficient cover (NMFS 2008). To date, approximately 5.6 km (3.5 mi) of habitat enhancement projects have been constructed, and their performance is continuously evaluated through an adaptive management process (Porter et al 2014). During the high-flow years of 2017 and 2019 several enhancement sites experienced significant changes, largely deposition of coarse gravel. Local managers theorize that this deposition is largely a result of tributary-supplied sediment being resuspended and transported downstream by sustained dam releases during flood risk management season (shown in conceptual model in Figure 5). Undammed tributaries provide a large sediment load to the middle and lower reaches of Dry Creek. This sediment load is redistributed by extended high flow releases from WSD during flood control season. The extended high flow releases are generally considered to have

sufficient energy to transport the sediment supplied by the tributaries, but not enough to mobilize the heavily armored bed along Dry Creek's mainstem. Newly constructed restoration projects are vulnerable to aggradation because of their larger cross-sectional area and corresponding reduced sediment transport capacity.

WSD – Dry Creek Conceptual Model

```
┌─────────────────────────────┐
│ Winter begins – several     │
│ storms arrive while WSD      │
│ below flood pool             │
└─────────────────────────────┘
        │
┌─────────────────────────────┐
│ LS rises above WSD flood pool│
└─────────────────────────────┘
        │
┌─────────────────────────────┐
│ WSD increases releases       │
└─────────────────────────────┘
        │
┌──────────────────────────┐          ┌──────────────────────────┐
│ Next storm, high inflows │─────────▶│ Tributaries supply water │
│ forecast to LS and       │          │ and sediment             │
│ elsewhere in basin       │          └──────────────────────────┘
└──────────────────────────┘                      │
        │                                          │
┌──────────────────────────┐                      │
│ WSD stops releases        │                      │
│ (flood control)           │                      │
└──────────────────────────┘                      │
        │                                          │
┌──────────────────────────┐                      │
│ Storm ends, LS still above│                      │
│ flood pool                │                      │
└──────────────────────────┘                      ▼
        │                          ┌──────────────────────────────┐
┌──────────────────────────┐       │ Sustained high flows in Dry  │
│ Increase releases, long   │──────▶│ Creek                        │
│ way to go to get below    │       └──────────────────────────────┘
│ flood pool                │                      │
└──────────────────────────┘                      ▼
                                   ┌──────────────────────────────┐
                                   │ Sediment transported away    │
                                   │ from tributaries confluences │
                                   │ down Dry Creek               │
                                   └──────────────────────────────┘
```

Figure 5. Conceptual model of how sustained high flows in Dry Creek lead to sediment transport away from tributary confluences and into restoration sites.

The restoration projects examined in this study are named, in order from upstream to downstream, Weinstock (WS), Truett-Hurst (TH), Ferrari Carrano Olson (FO), and City of Healdsburg (CH). The WS site was constructed in 2018 and includes 1 side channel excavated into the right floodplain of Dry Creek. The TH site was originally constructed in 2016, and included a relatively long side channel excavated into the left floodplain of Dry Creek. It experienced significant channel changes (deposition and realignment) during the high flow water years of 2017 and 2019. The FO site was originally constructed in 2018, and included a long side channel in the left floodplain, and shorter one on the right. It experienced significant deposition and realignment following water year 2019. The CH site was constructed in 2017 with a side channel excavated into the left bank, and also experienced significant deposition in water year 2019.

While the TH, FO, and CH sites have all shown signs of meeting ecological goals, they have needed significant and somewhat costly adaptive management repairs to continue to provide salmonid rearing habitat following large water years with extensive flood control releases. This study examines the effects of changing flood control releases in water year 2019, which is the only year for which there was before-and-after data for all restoration sites as well as extensive flood control releases, on salmonid rearing habitat and sediment deposition both within the restoration sites and throughout Dry Creek.

4. Methods

4.1. Overview
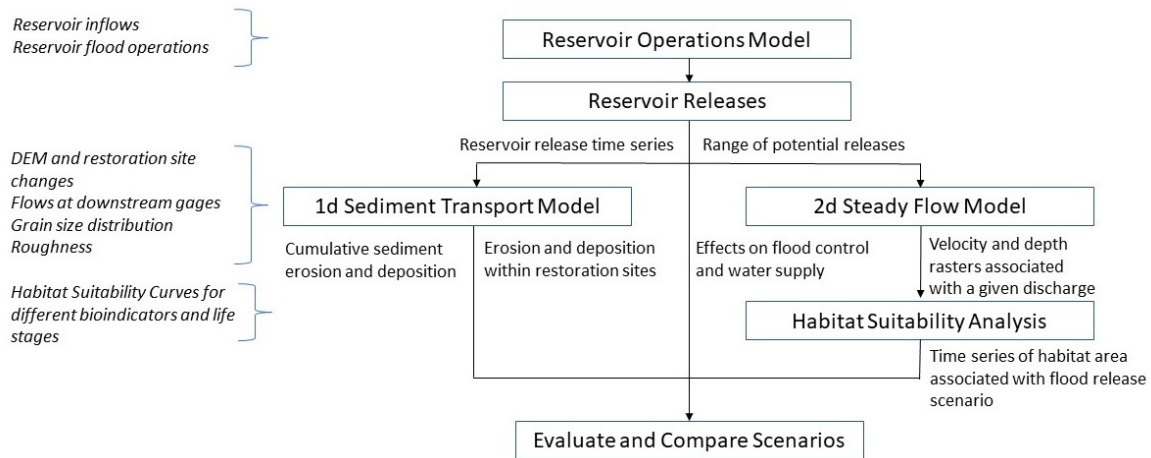
Figure 6. Schematic of methods and data inputs used in this study.

Several approaches were used to evaluate the impact and benefits of changing reservoir releases during the flood season (Figure 6). A mass balance reservoir simulation model with an hourly time step was used to evaluate benefits to water supply and flood risk management. A one-dimensional (1D) morphodynamic model was used to evaluate the effects on channel change throughout the river corridor. A two-dimensional (2D) steady state hydraulic model was combined with habitat suitability curves to evaluate the effects on endangered salmonid rearing habitat within the restoration sites and elsewhere in the river. Goals were identified for each analysis, but it should be noted that it may not be possible to

meet all goals simultaneously. Data inputs for this analysis included detailed annual topographic surveys

of restoration sites, dam inflow and outflow time series, river flows from downstream gages, grain size

distributions from a series of pebble counts, and some components (e.g. roughness values) from past

modeling efforts. These analyses, as well as the specific hypotheses and tests within each one, are

summarized in Table 2.

Table 2. Analyses, goals, hypotheses, and comparative tests applied to each scenario in this study.

Analysis	Goal	Hypothesis	Test with respect to current operations
Reservoir Operations	Do not negatively impact reservoir operations in the context of water supply and flood control	Modified reservoir releases will not be any worse than actual releases at meeting flood control objective, and may have ancillary water supply benefit	Comparison of peak storage, final storage, date of peak storage, peak flow at Geyserville
Sediment Transport	Less deposition in Restoration Sites	Less time above sediment transport threshold of ~ 42 m^3/s (1500 cfs) will result in less scour and sediment transport river-wide, which will result in less deposition in restoration sites	Less/more deposition is defined as at least a 20% difference relative to modeled actual flows
	More habitat development elsewhere in river	Less time above sediment transport threshold of ~ 42 m^3/s (1500) cfs will result in less scour and sediment transport river-wide, and as a result retain more sediment near tributary confluence zones, thereby developing more habitat	Less/more deposition is defined as at least a 20% difference relative to modeled actual flows
Habitat Suitability	Fry rearing habitat availability	Lower magnitude flood releases during fry rearing months of March-June will increase duration of habitat availability	Comparison of fry habitat acre-days from March to June, within restoration sites and whole river
	Juvenile rearing habitat availability	Lower magnitude flood releases will increase duration of habitat availability for rearing juveniles during flood release season	Comparison of juvenile habitat acre-days from December to June, within restoration sites and whole river

4.2. Reservoir Operations Model

A mass balance reservoir simulation model was built to compare current (baseline) and alternative water management strategies operation for comparing their effects on water supply and flood management. The simulation model calculates storage at any given time step as equal to the storage at previous timestep plus inflow minus outflow at that time step. WSD does not have an inflow gage, and so the only available inflow values are calculated based on daily storage and daily average reservoir release. Hourly calculated inflow values were also available, but were not used because of unacceptable error in the data. Evaporation was not included in the model as it would affect all scenarios relatively similarly. Experimental releases were obtained by modifying the actual (i.e. baseline) flood control releases to achieve different scenarios. Experimental release results were then compared to the baseline release results in terms of their performance in water supply (final storage) and flood control (peak storage, date of peak storage, and peak flow relative to the 198 m^3/s (7,000 cfs) limit at Geyserville).

The primary test case for this study was water year 2019. Topographic change data from all four sites were available for this year, and it saw extended high flow releases during the winter and spring months. Reservoir releases greater than or equal to 28 m^3/s (1,000 cfs) were rounded to the nearest thousand to simplify analysis. In this context, flood control releases were defined as releases of 28 m^3/s (1,000 cfs) or greater for a duration of 5 hours or longer. Three different experimental dam release scenarios were initially defined, and a fourth was later added based on results from habitat suitability modeling.

The scenarios were initially designed to minimize time spent above the sediment transport threshold for bed material in the stream below Pena Creek, typically taken to be around 42 m^3/s (1,500 cfs) (Interfluve Inc. 2013). This criterion was hypothesized to promote habitat development near tributary confluences and reduce deposition of sediment within the constructed restoration sites. A potential disadvantage of this criterion is that it requires more time with reservoir storage spent in the flood pool during high runoff months.

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The first scenario set the release to be the maximum controlled dam release, 170 m³/s (6,000 cfs), and then shortened the duration so that volume released during the release episode matched that released in the baseline scenario. Following this, a baseflow of duration equal to the old release duration minus the new release duration was inserted back into the time series following the flood control release to preserve the timing of future releases. The second scenario set the release to 28 m³/s (1,000 cfs), intended to be below the transport threshold. By definition, none of the releases decreased in magnitude, so their duration was either extended or stayed the same. The resulting series of releases ended up being longer than one year, so it was shortened for comparison to the other scenarios. The third scenario used outflows from the first during the winter (Dec-Jan-Feb), and then outflows from the second during spring/summer (Mar and later). The last experimental scenario release was set to be 57 m³/s (2,000 cfs), which was the discharge that led to maximum rearing habitat area within the constructed restoration sites. To evaluate bookends of dam operations, two more scenarios were added, one where the outflow is equal to the inflow, and another where the outflow is set to be the 3 m³/s (100 cfs) baseflow for the entire modeling period. These scenarios are summarized in Table 3.

Table 3. Dam release scenario definitions used in this study.

Scenario Number	Scenario Name	Peak outlow (m³/s)	Description
1	Baseline	142	Actual reservoir releases from WY2019
2	6k	170	All flood releases converted to 170 m³/s (6,000 cfs) magnitude and corresponding shorter duration to equal same volume of actual extended releases
3	1k	28	All flood releases converted to 28 m³/s (1,000 cfs) magnitude and corresponding longer duration to equal same volume of actual extended releases
4	Hybrid	170	Scenario 2 for Dec-Feb, and Scenario 3 for Mar-Jun
5	2k	57	All flood releases converted to 57 m³/s (2,000 cfs) and shifted duration based on results of habitat suitability within restoration sites
6	Inflow	380	Hourly outflow is set to be average daily inflow

Scenario Number	Scenario Name	Peak outlow (m^3/s)	Description
7	Baseflow	3	Outflow is set to 3 m^3/s (100 cfs) baseflow

4.3. Hydraulic Modeling

Two models were built in the Hydrologic Engineering Center River Analysis System version 6.0 (HEC-RAS) for the project area, one was a 1D sediment transport model and the other was a 2D hydrodynamic model (Brunner 2002). These models were based on previous models put together for the restoration projects, and modified to meet the needs of this study. Flows obtained from the gages were used in season-long simulations to match observed topographic changes in the restoration projects. The topography data in the models come from a mixture of airborne LiDAR data (WSI 2013), Unmanned Aerial Vehicle Structure from Motion, and high-resolution annual topographic surveys from four of the constructed restoration projects (Martini-Lamb and Manning, in preparation-a and Martini-Lamb and Manning, in preparation-b). The hydroflattened Digital Elevation Model (DEM) obtained from the LiDAR survey used had an average accuracy of 0.05 m and a ground density of 2.88 points/m^2 (WSI 2013). The restoration site surveys and DEMs derived from them have an average horizontal accuracy of <0.05 m, an average vertical accuracy of <0.02 m, an instream point density between 0.23 and 0.45 points/m^2, and an overall point density between 0.06 and 0.74 points/m^2 (Mark Goin, Sonoma Water, pers. comm.). For both models, the inflow boundary conditions consist of the dam release and tributary flow, and the outflow boundary conditions used the normal depth approximation.

4.3.1. Flow Data

There are four currently running flow gages on Dry Creek, which have been operating for various amounts of time. One of them, USGS station #11465200, has been in continuous operation since before the dam was built and allows for a comparison of pre and post-dam flow duration curves. This gage also

measured flows of sediment for several years around the time of dam construction. The post dam sediment data was used as a starting point for sediment rating curves.

For flows, the gages on Dry Creek along with the dam release gage were used in a mass balance approach to separate flow from the tributaries and from the dam. The tributary flow was distributed by watershed area, and the tributaries are grouped into upper and lower groups based on where they are in the watershed (Gianfagna et al 2015). For the purposes of this hydraulic model, all flow was assumed to come either from the dam or from the 12 named tributaries of Dry Creek.

4.3.2. Geomorphic Change Analysis

Within the case study area, Sonoma Water has been carrying out annual topographic surveys at each restoration site with a combination of ground based total station surveying and structure from motion imagery using an unmanned aerial vehicle. These detailed data were provided for four sites throughout the watershed and give a topographic snapshot before and after each winter. A Geomorphic Change Detection (GCD) tool in ArcMap was used to determine the topographic change between successive monitoring episodes before and after the wet season of water year 2019 (Wheaton et al 2010a, Wheaton et al 2010b). This was then used for calibration of the sediment transport model.

4.3.3. 1D Morphodynamic Model

The 1D sediment transport model simulates a quasi-unsteady flow through a series of cross-sections based on the project topographic data that make up the modeling domain (Brunner and Gibson 2005). The 1D sediment transport model compares sediment supply to transport capacity on a cross-sectional basis and applies a veneer of erosion or deposition across all wetted nodes (Gibson and Nelson 2016). While 1D sediment transport models have some drawbacks in representing all of the various morphodynamic processes going on in a river, they have been shown to be effective at making at broad stroke predictions over long time periods and over large spatial extents (Formann et al 2007, Sanyal et al 2021). The model was tested with different sediment transport functions, but the final used for

experimental comparisons was the Laursen-Copeland sediment transport formula, which uses an excess shear calculation and empirical correction to model transport of each grain size (Copeland 1993). The Laursen-Copeland formula is applicable for a wide range of grain size distributions (Gibson 2011). The water inflows were based on the gage analysis described above, and normal depth was used as the downstream boundary condition.

The initial grain size distribution in the channel was based on 28 reach-averaged pebble counts collected by Interfluve, Inc in 2010 as part of the study work for the habitat restoration projects, along with several additional pebble counts collected as part of this study. Flow-load relationships in the absence of measured data are difficult to predict, but as the tributaries are not supply-limited relative to the mainstem of Dry Creek, the incoming grain size distribution was assumed to have a mean value in the medium/coarse gravel range, which is significantly finer than existing bed material (Gibson and Cai 2017). As the focus of this component of the study was transport of coarse-grained bed load, materials finer than sand were not included in the model and the dam was assumed to not release any sediment, only water.
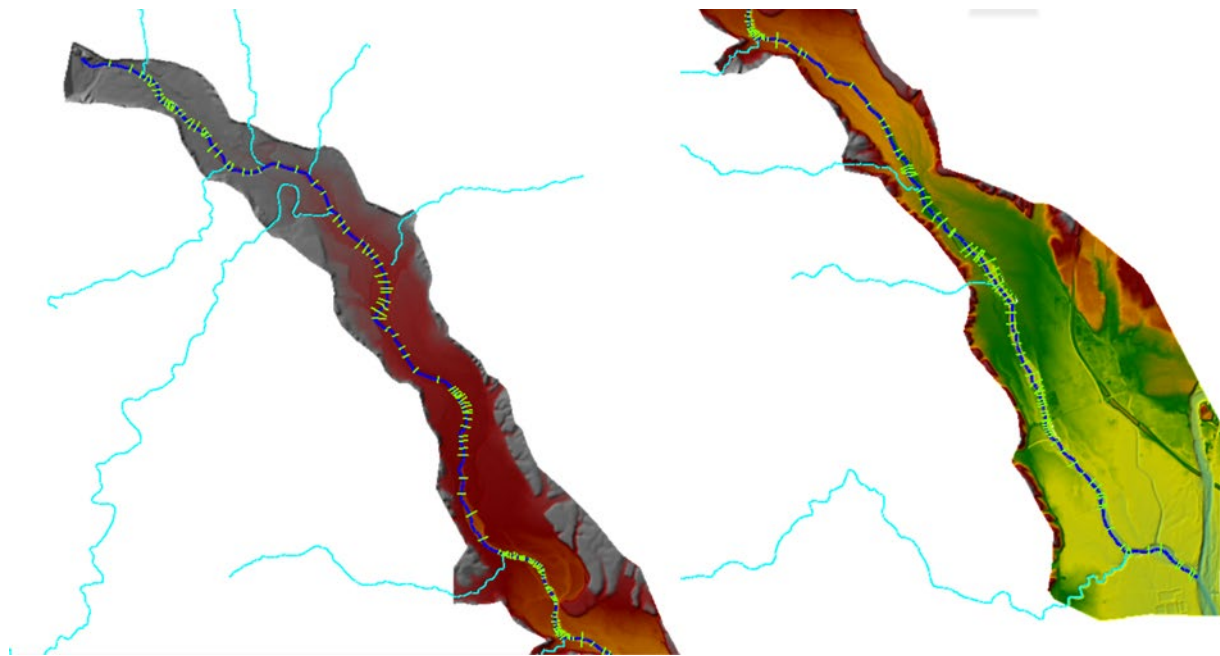

Figure 7. Modeling domain with cross sections shown in green and tributaries shown in light blue. Left-hand side is the upstream portion of the model, and right-hand side is the downstream portion.

The modelling domain was represented by 213 cross sections based on terrain that was a combination of pre-project LiDAR and detailed topographic surveys collected as part of post-project monitoring (Figure 7). Cross section spacing was varied such that areas of interest had a higher density of cross sections. Grain sizes were based on previously collected pebble counts, and incoming tributary grain sizes were assumed as described above (Figure 8). Interpolation was used to supply the model with grain size distribution data in cross sections without pebble counts. The majority of cross sections had a median grain size (d50) within the coarse gravel range (16-32 mm). The assumed incoming sediment supply had a d50 of 4 mm, which is between very fine gravel and fine gravel. The assumption of a sediment load with a finer gradation than the existing bed is in line with other models and recommendations in the HEC-RAS user's manual, but this is one of the most uncertain model parameters (Gibson et al 2017).

The parameters of the flow-load rating curve for the tributaries were calibrated to match observed and modeled net sediment flux within restoration sites. To minimize the degrees of freedom within the model, all tributaries were modeled as lateral inflows of sediment and water, and assumed to have the same flow-load relationship. The initial flow-load relationship was based on a power regression of the flow-load values available for several years after dam closure. Following this, the exponent and coefficient of the flow-load relationship were varied in an attempt to match the modeled longitudinal cumulative volume change within the restoration sites to the volume change calculated in the GCD analysis for each site (similar to Gibson 2011).
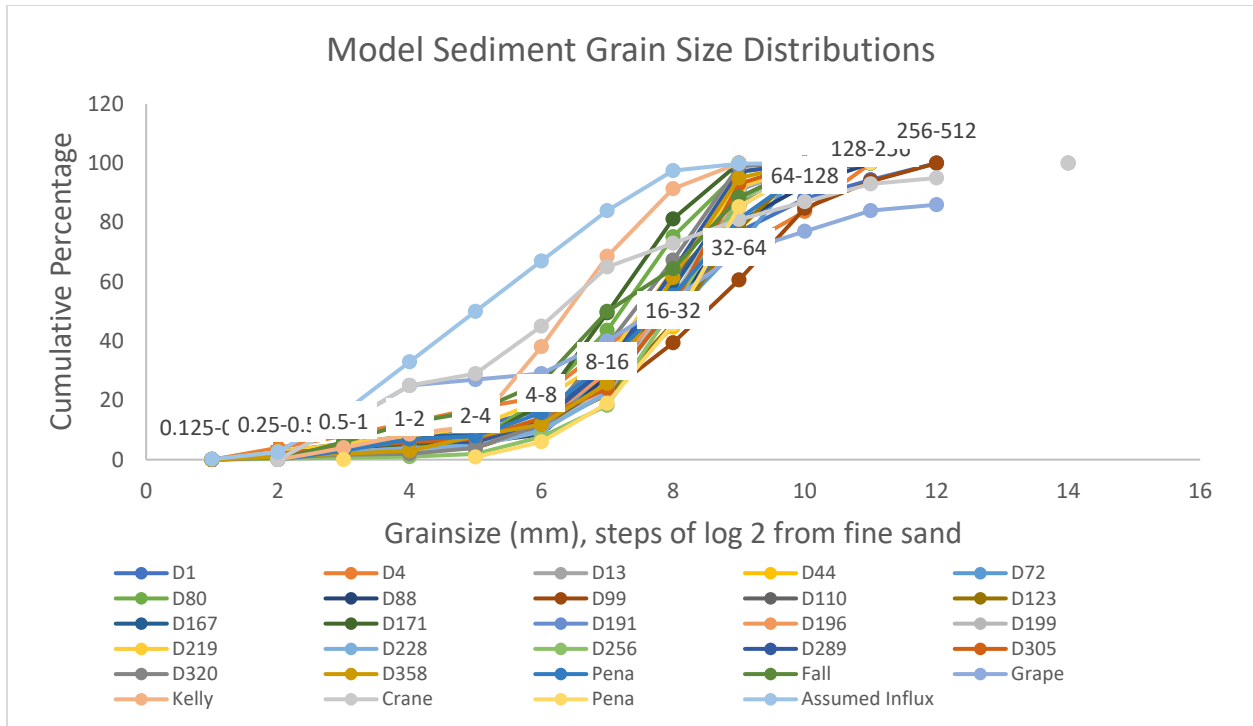
Figure 8. Grain size distributions used within the 1D morphodynamic model. Tributary bed gradation not used in this model, but shown for reference.

Longitudinal cumulative volume change was used for relative comparisons in geomorphic change between the scenarios. Volume change within restoration sites and tributary confluence zones was evaluated relative to the changes shown within the actual flow scenario. Deposition within tributary confluence zones was evaluated because of these zones' outsize importance in providing physical habitat heterogeneity and supporting biodiversity, especially in dammed rivers (Rice et al 2006). Tributary confluence zones were defined for the 5 tributaries (Schoolhouse, Dutcher, Pena, Grape and Mill) with a watershed area greater than 7.8 km^2 (3 mi^2). Tributaries with smaller watersheds were assumed to not be geomorphically significant because of the comparatively large size of the Dry Creek watershed (Benda et al 2004). To scale with watershed area, these zones were defined such that the length of the zone downstream of the confluence in feet was equal to 100 times the watershed area in square miles. Because of the high uncertainty within the model, 20% was selected as the threshold to define more or less deposition/erosion relative to actual flows.

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At the conclusion of the attempted calibration, model performance was varied between the restoration sites because of various factors described in the Limitations section below. Because the model still had large discrepancies between observed and modeled channel change after calibration, it should only be considered for providing relative or comparative results between the dam release scenarios (and even this only upstream of the limits of Russian River backwater influence near Westside Road bridge), not predictive results that would be accurate in terms of volume. This is similar to other studies that take a numerical experiment approach to illustrate differences in habitat response to different discharges (e.g. Lane et al 2018).

4.3.4. 2D Steady Flow Model

The 2D hydrodynamic model solves the shallow water equation approximation of the Saint Venant equations in a non-orthogonal finite element grid laid over the modeling topography. Within HEC-RAS, breaklines were used to align the computation points with the dominant flow directions, and refinement regions were used to reduce point density on the vineyard terrace level because it does not get inundated during controlled dam releases (Figure 9). The roughness values used were not specifically calibrated for this model, and instead came from a model that had the purpose of predicting inundation area and water surface elevations during the theoretical 100-year storm.

The 2D model was used in a steady-state configuration (HEC-RAS does not have a 2D steady state option, but each discharge was run for days at a time to allow flow to stabilize throughout the model domain) along with habitat suitability curves for the species of interest to create discharge - habitat area rating curves for the entire river corridor. The 2D model only represents wetted area as a result of dam releases and ignores tributary inputs. During winters with extensive flood releases, the tributary flow pulses are transitory in nature, on the order of hours or days, while dam releases can last for weeks. Similar to the 1D model, this model used normal depth as a downstream boundary condition. The non-orthogonal mesh used was user generated and contained approximately 45,000 cells that ranged in size between 9.5 m^2 (102 ft^2) and 5,853 m^2 (63,000 ft^2), with an average cell size of 186 m^2 (2,000 ft^2). The

model was run with a Courant-number-based variable time step and the Eulerian-Langrangian momentum approximation of the full shallow water equations.
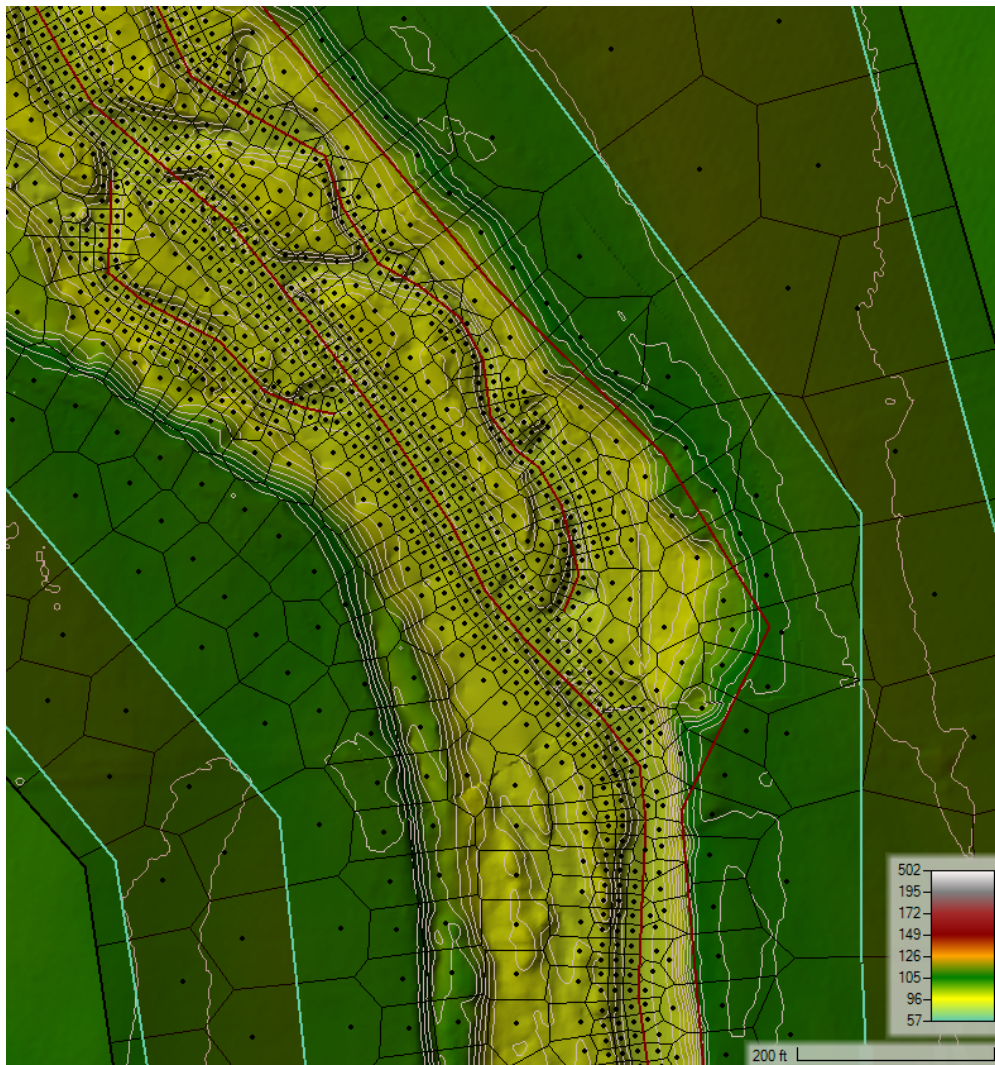
Figure 9. Model mesh grid and computation points, shown on underlying terrain with 2 ft contours. Breaklines (maroon lines) used to align computation points with flow and refinement regions (shaded olive areas) used to reduce point density on infrequently inundated areas. Map units in ft from unit system used in HEC-RAS.

The 2D steady flow model provided depth and velocity outputs for a range of discharges that spanned the range of flows within the dam release scenarios. The range of flows was between 3 and 396 m^3/s (100 and 14000 cfs). Each flow was run through the model for 3 days, with smooth transitions between. The model had an overall volume accounting error of less than 0.001%. Depth and velocity profiles were exported from the hydraulic model and resampled in a GIS to a 0.91 m (3 ft) cell size raster grid.

4.4. Habitat Suitability Analysis

The habitat suitability curves used for evaluating the winter and spring rearing habitat potential for Coho salmon (*Oncorhynchus kisutch*) and steelhead (*O. mykiss*) were developed by the California Department of Fish and Wildlife in a nearby tributary of the South Fork Eel River (Gephart et al 2020). The curves are derived from a histogram of actual habitat utilization based on field data (Bovee et al 1998). Depths and velocities are set into bins and the histogram represents usage in each depth/velocity class. The histograms are then normalized (i.e. highest value set to 1, and all other values scaled proportionally) and a curve is generated using a smoothing function. This survey had HSCs for both steelhead and coho, and separated them by fish less than 6 cm and greater than 6 cm, corresponding to fry and juvenile age classes, respectively. Peak velocity suitability values were found in pools of slow moving water, and were less than 0.15 m/s (0.5 ft/s) for all species and life-stages except steelhead juveniles. Peak depth suitability values for all species and life-stages were between 0.15 and 0.61 m (0.5 and 2 ft). These peak suitability values are comparable to the design criteria used for the habitat enhancement projects (Figure 10).

For this study, the curves were used alongside depths and velocities exported from the 2D model to generate areas of suitable habitat for each species and lifestage. This analysis was conducted using River Architect, a free open-source software that allows for automated ecohydraulic analyses of varying discharges (Schwindt et al 2020). The software takes depths, velocities, topography, and grain size data as inputs. It converts the habitat suitability curves to piecewise linear functions and applies them to the hydraulic rasters for each discharge. A combined habitat suitability index raster was generated from the geometric mean of depth and velocity suitability rasters, and then a threshold of 0.5 was chosen to define areas of suitable habitat (Leclerc et al 1995). River Architect then produced curves of discharge vs habitat area for each species and lifestage both within the restoration sites and for the whole river corridor.
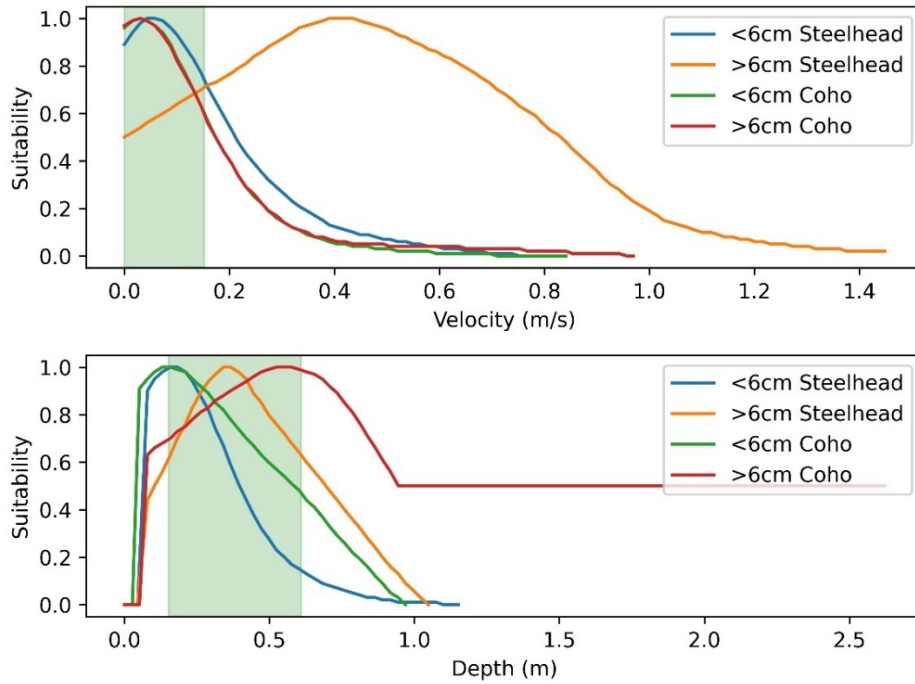
Figure 10. Rearing habitat suitability curves from Gephart et al 2020. Green shaded areas indicate design targets for fry used in enhancement projects.

4.5. Release Scenario Comparison

The discharge-habitat area curves were applied to the various dam release scenarios to generate comparable numbers of duration of habitat availability during the model duration. The trapezoidal integral approximation was used to generate cumulative duration of habitat availability for Coho and steelhead fry and smolt for each scenario both in the restoration sites and in the entire 2D model domain. The scenario comparison approach used here is somewhat similar to the structured decision making approach presented in DeWeber and Peterson (2020), although it has fewer habitat variables and the added component of 1D sediment transport modeling (DeWeber and Peterson 2020).

Along with habitat performance, the different scenarios were compared in terms of their effects on channel change within the restoration sites and tributary confluence zones. The reservoir simulation model was used to evaluate the water supply and flood risk management performance of the different scenarios. The scenarios were evaluated using the criteria defined in Table 2 to determine how well they met objectives in all three modeling methodologies.

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5.Results

5.1.Reservoir Operations Model

The reservoir mass balance simulation model allowed for comparison of water supply and flood control functions between different release scenarios. All scenarios started with the actual initial storage in water year 2019 (242 million m^3 (196 thousand acre-feet (TAF))), and ended with varying amounts of storage. The inflow scenario had the least amount of water stored at the conclusion of the water year, while the baseflow scenario had the most. Of the operationally possible scenarios, the 1k scenario was the most beneficial from a water supply perspective and had the most water at the end of the season at 291 million m^3 (236 TAF). This is approximately 11 million m^3 (9 TAF) more than the baseline scenario. The 1k scenario also encroached furthest into the flood control pool of the reservoir, with a peak storage on March 17 of 430 million m^3 (349 TAF), which is 39 million m^3 (32 TAF) from the spillway crest storage of 470 million m^3 (381 TAF).

Table 4. Reservoir simulation summary table. All values except date expressed as percent change relative to Baseline.

Scenario	Final Storage	Peak Storage	Date of Peak Storage	Total Volume Released	Peak Flow	Peak Flow at Yoakim Bridge
Q_out (Baseline)	281 x 10^6 m^3	378 x 10^6 m^3	3/2/2019	346 x 10^6 m^3	142 m^3/s	238 m^3/s
6k	-6%	-1%	3/2/2019	+5%	+20%	0%
1k	+4%	+14%	3/17/2019	-3%	-80%	+7%
hybrid	-10%	+4%	3/17/2019	+8%	+20%	0%
2k	0%	+13%	3/17/2019	0%	-60%	+19%
inflow	-14%	-36%	10/4/2018	+11%	+168%	+155%
baseflow	+91%	+49%	6/6/2019	-74%	-98%	-3%

On the flood risk management side, flows exceeded the 198 m^3/s (7,000 cfs) limit at the Geyserville gage across all scenarios during the late February storm, with scenario 6 having the highest peak flow (607 m^3/s (21,448 cfs)). With the exception of this storm event, the non-inflow release scenarios stayed under the limit for the duration of the simulation. Figure 11 and Table 4 show a summary comparison of the various scenarios.
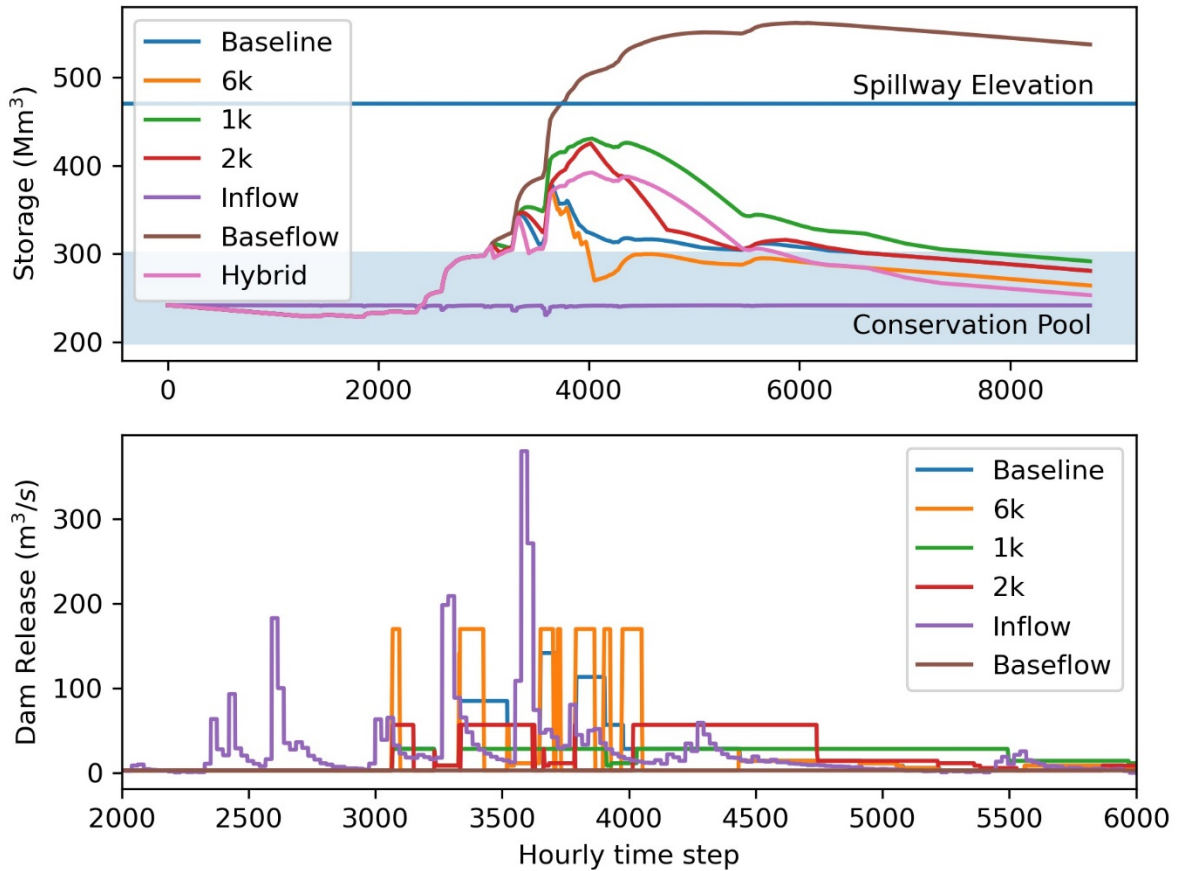

Figure 11. Storage and dam release time series for all scenarios.

5.2. Hydraulic Modeling

5.2.1. GCD analysis

The GCD analysis for water year 2019 showed deposition at all 4 of the restoration sites (Figure 12). The FO site, the largest of the four, experienced the largest net volume of deposition, ~ 14,500 m^3

(19,000 yd^3). This site also had the most deposition relative to its length. The WS site, the smallest of the four and also with the least amount of tributary influence, experienced minimal change, ~ 350 m^3 (460 yd^3) of deposition.
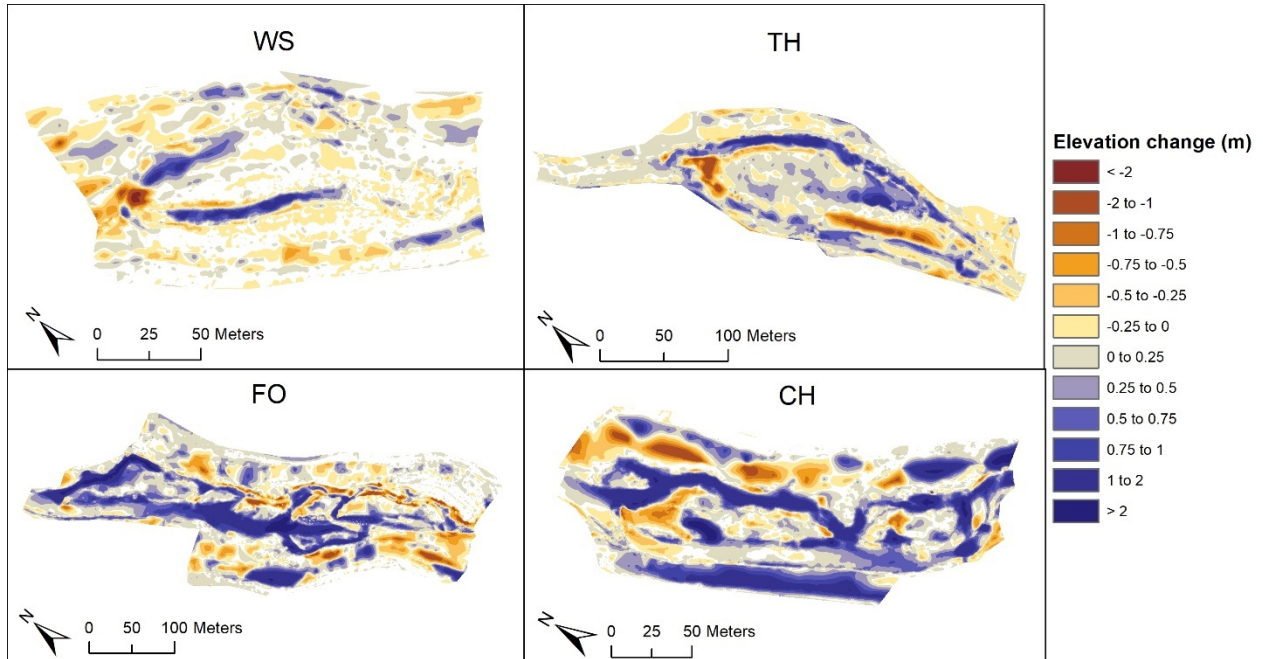
Figure 12. GCD results from water year 2019.

5.2.2. 1D Morphodynamic Model

The 1D morphodynamic model produced estimates of channel change for water year 2019. The flow-load function for tributary sediment supply was modified as part of the calibration process to try to match deposition within the restoration sites to what was observed during this water year. The various model runs were inconsistent when compared to the observed GCD results in the restoration sites, with average percent error ranging between 521% and 649%. The large percent errors are largely due to overprediction of deposition in the WS and TH sites, and overprediction of erosion in the CH site. The model performed better for the FO site, with percent errors ranging between 7% and 18% for the different calibration runs.

Within the restoration sites, the calibration runs were all significantly closer to each other than to the observed data (Figure 13). Peak flows and water surface elevations matched observed values at the

flow gages much more closely than sediment deposition in the restoration sites. Pebble counts conducted for this study within the restoration site deposits had d50s within the range of coarse gravel (TH side channel and FO former main channel) or medium gravel (FO side channel), which matched the modeling results.
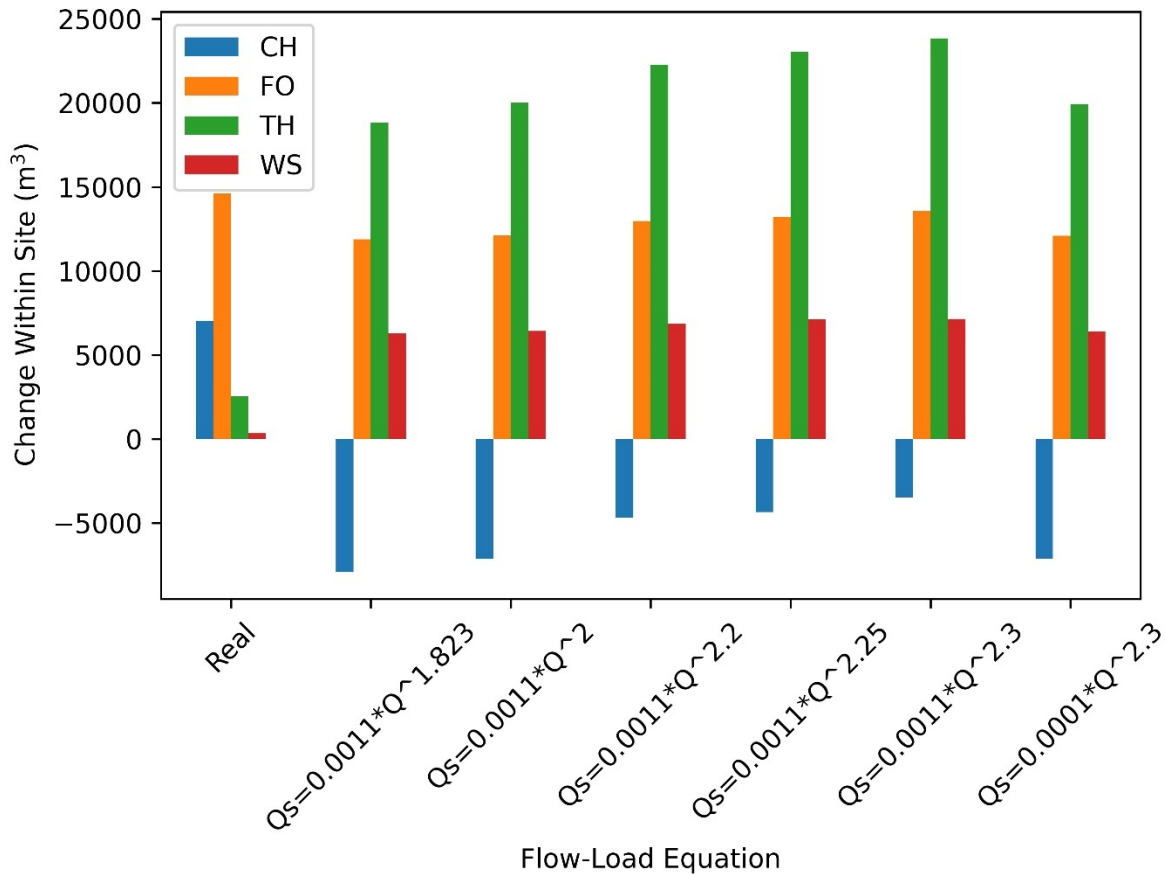
Figure 13. Channel change within restoration sites for calibration runs compared to observed values.

The longitudinal cumulative volume change curves show a larger difference between calibration runs (Figure 14). Some of the runs had net erosion, while others had net deposition. The runs with net deposition show large inflection points at Pena and Mill Creeks. The large cobble (128-256 mm) and small boulder (256-512 mm) grain sizes did not move at all between cross sections, and the small cobble (64-128 mm) class experienced only minimal change.
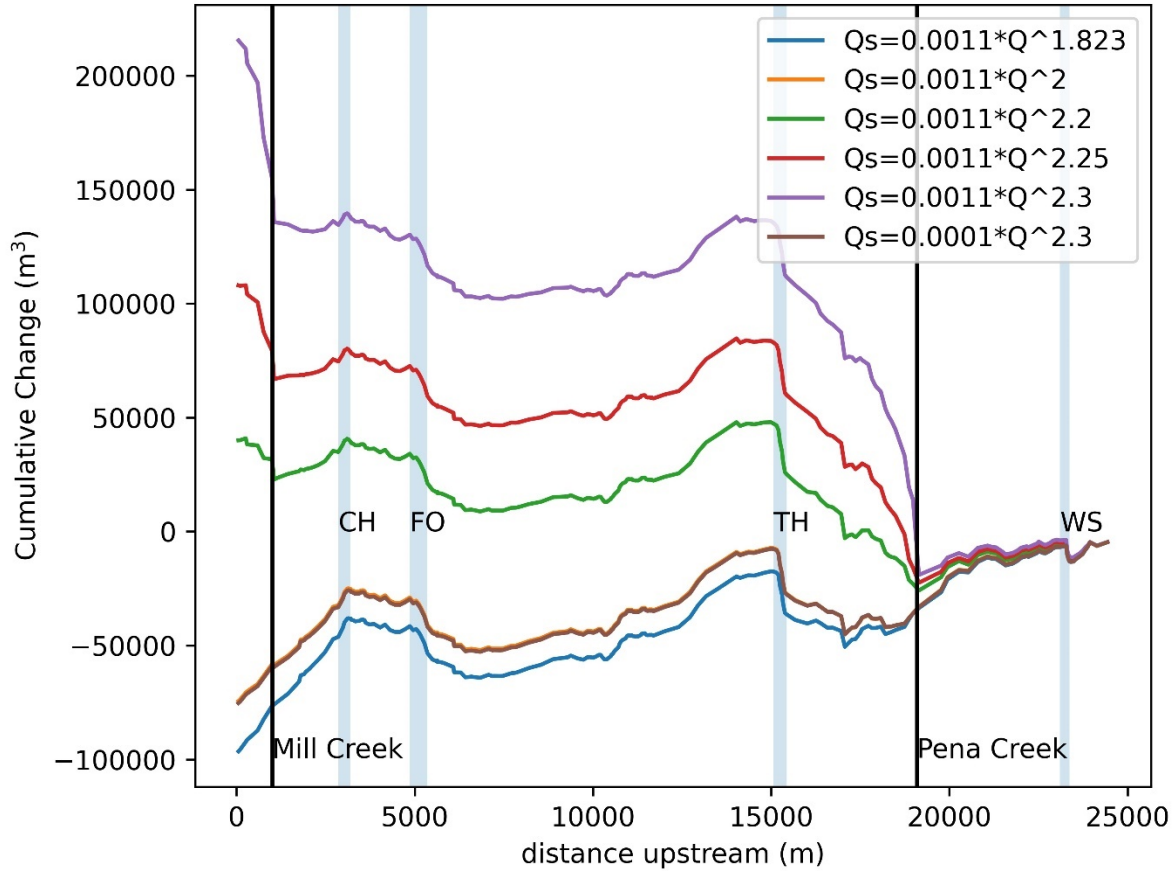
Figure 14. Longitudinal cumulative volume change for calibration runs summed from upstream to downstream and plotted from downstream to upstream. Blue bars indicate extent of restoration sites and black lines indicate two largest tributaries.

The flow-load relationship of $Qs = 0.0011 * Q^{2.2}$ (where Qs is volumetric discharge of sediment and Q is volumetric discharge of water) was selected as the function to be used in comparison of the experimental dam outflow scenarios. This function showed a slightly erosional regime upstream of Pena Creek, and somewhat depositional below, especially in the TH and FO restoration sites and the area immediately downstream of Pena Creek. This function was selected because it had the lowest percent error of scenarios that showed net erosion upstream of Pena Creek and net deposition below.

The dam release scenarios described in the reservoir simulation model section above were run through the 1D morphodynamic model to evaluate their relative effects on channel change within the restoration sites and throughout the river corridor (Figure 15). There was a wide range in their effects of both aspects of the river's geomorphological regime. In general, scenarios with higher peak flows had less

net aggradation throughout the river corridor and more deposition in the restoration sites that the model showed deposition in (CH had erosion in all model runs). Similar to the calibration runs, the model showed large inflection points where the largest tributaries joined Dry Creek. Total sediment out ranged between 79,500 m^3 (104,000 yd^3 (baseflow)) and 231,000 m^3 (302,000 yd^3 (inflow)). Of the operational scenarios, the 1k cfs scenario had the smallest amount of sediment exported from the river corridor (115,000 m^3 (151,000 yd^3)).
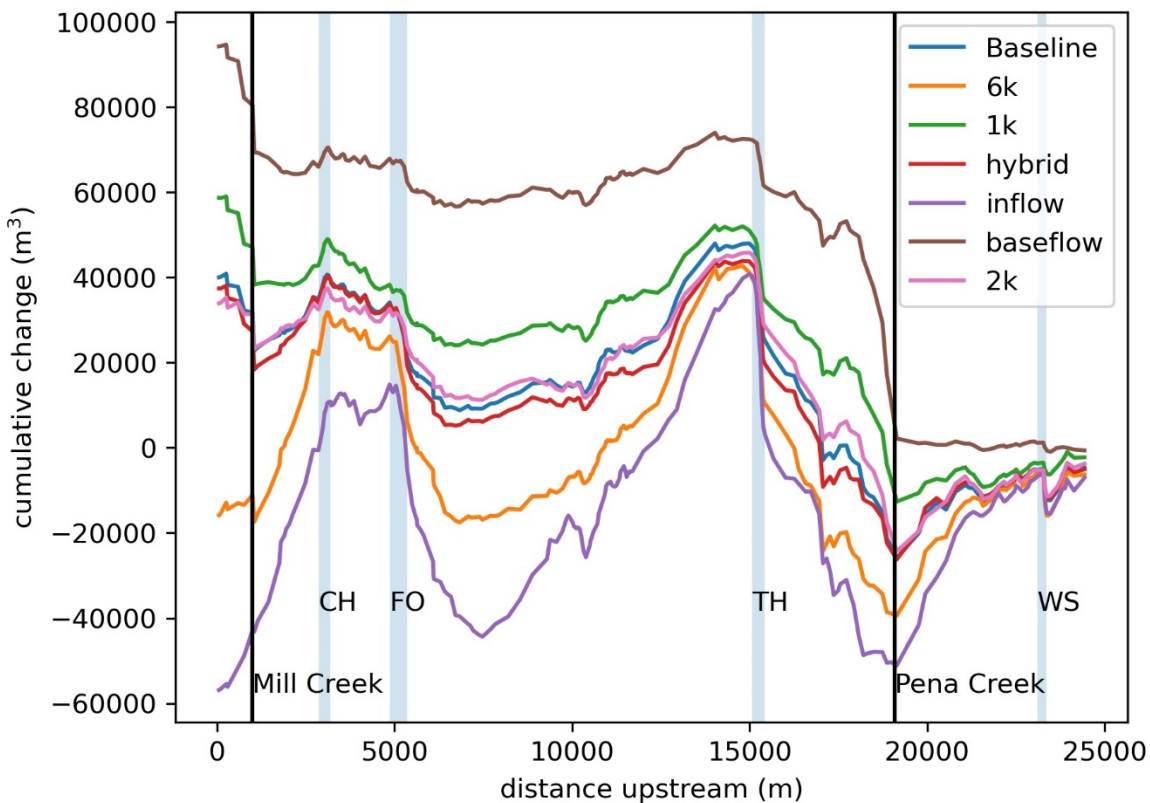
Figure 15. Longitudinal cumulative volume change for different dam release scenarios summed from upstream to downstream and plotted from downstream to upstream. Blue bars indicate extent of restoration sites and black lines indicate two largest tributaries.

Within the restoration sites and tributary confluence zones, relative patterns of channel change were similar to the longitudinal view. All scenarios showed the same direction of channel change as the calibration runs, but the magnitude of channel change varied significantly (Figure 16). As compared to the actual releases, the 6k scenario had approximately 30-40% more channel change within each site, while the 1k scenario had approximately 25-50% less channel change within each site. The 1k and

baseflow scenarios retained the most sediment within tributary confluence zones, with the 1k scenario having 51% more sediment retained in these zones. As above, the inflow and baseflow scenarios had the largest difference relative to the actual releases. The majority of the sediment deposited within the restoration sites was within the gravel size fraction.
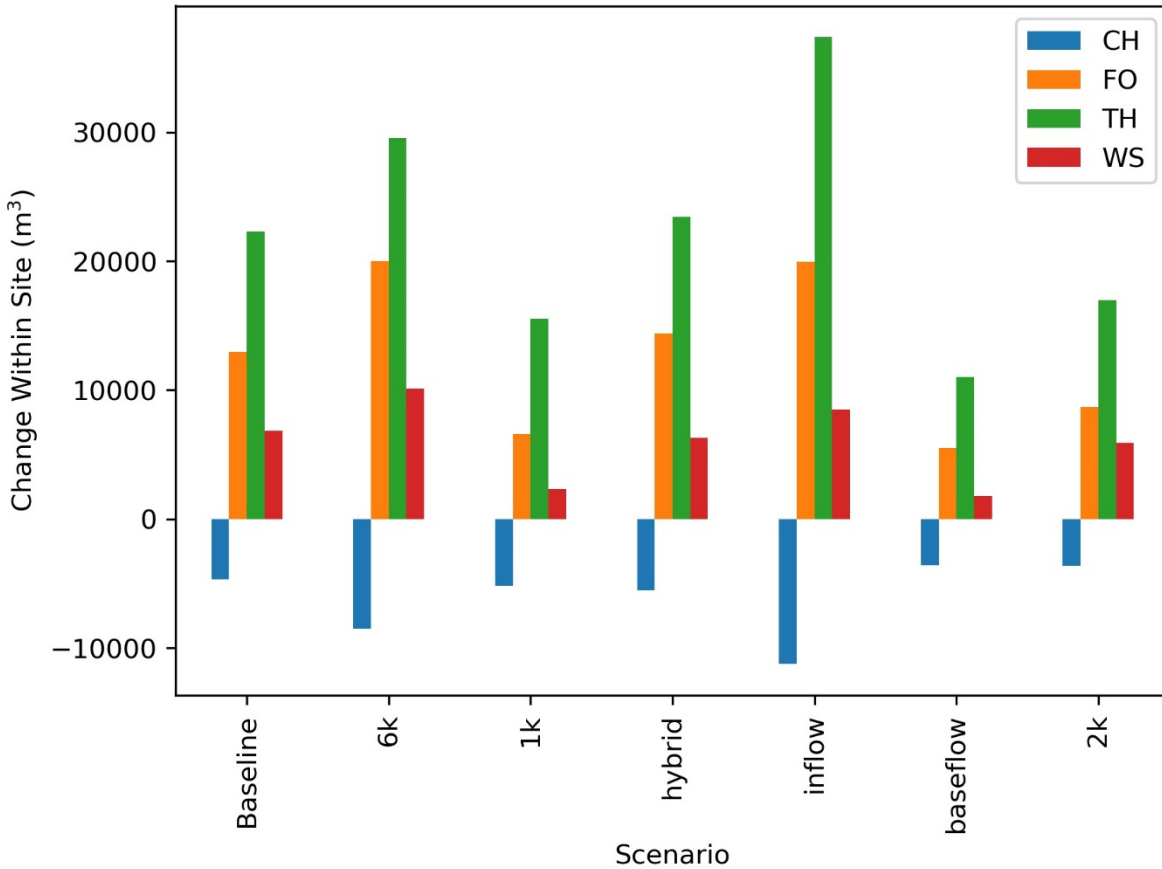
Figure 16. Channel change within restoration sites caused by alternative dam release scenarios.

5.3.2D Habitat Suitability

For the habitat suitability analysis within the restoration sites, all of the discharge-habitat area curves had a maximum area at 57 m^3/s (2,000 cfs) (Figure 17). For the whole river, maximum habitat area was at 396 m^3/s (14,000 cfs) due to breakout flooding on the vineyard terrace level, but for implementable flows generally at 85 or 113 m^3/s (3,000 or 4,000 cfs) (Figure 18).
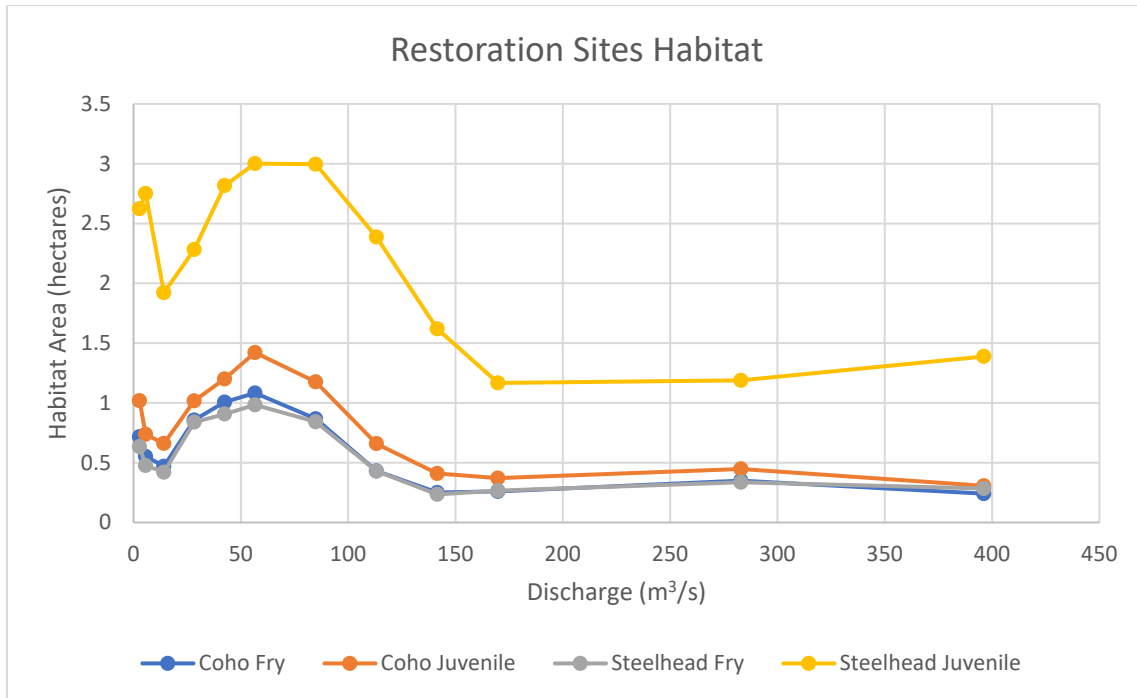
Figure 17. Discharge - Habitat Area curves for the restoration sites.

The discharge-habitat area curves showed differences for all 8 combinations of species, life stage, and restoration site vs. whole river habitat. The resulting habitat area time series were then integrated between March 1 and June 30 for fry, and December 1 and June 30 for juveniles to show the difference in cumulative habitat provided under each scenario (Table 5). Within the restoration sites, the 1k and hybrid scenarios performed best for fry of both species, while the baseflow scenario provided the most habitat for juveniles of both species. For the whole river, the 1k and hybrid scenarios also provided the most habitat for fry of both species, while the 1k scenario alone had the most habitat for Coho juveniles and the baseflow scenario provided the most habitat for steelhead juveniles. The differences between the actual flow scenario and the 1k scenario were largest for fry rearing of both species, with the 1k scenario representing a 16-32% improvement during the fry rearing period.
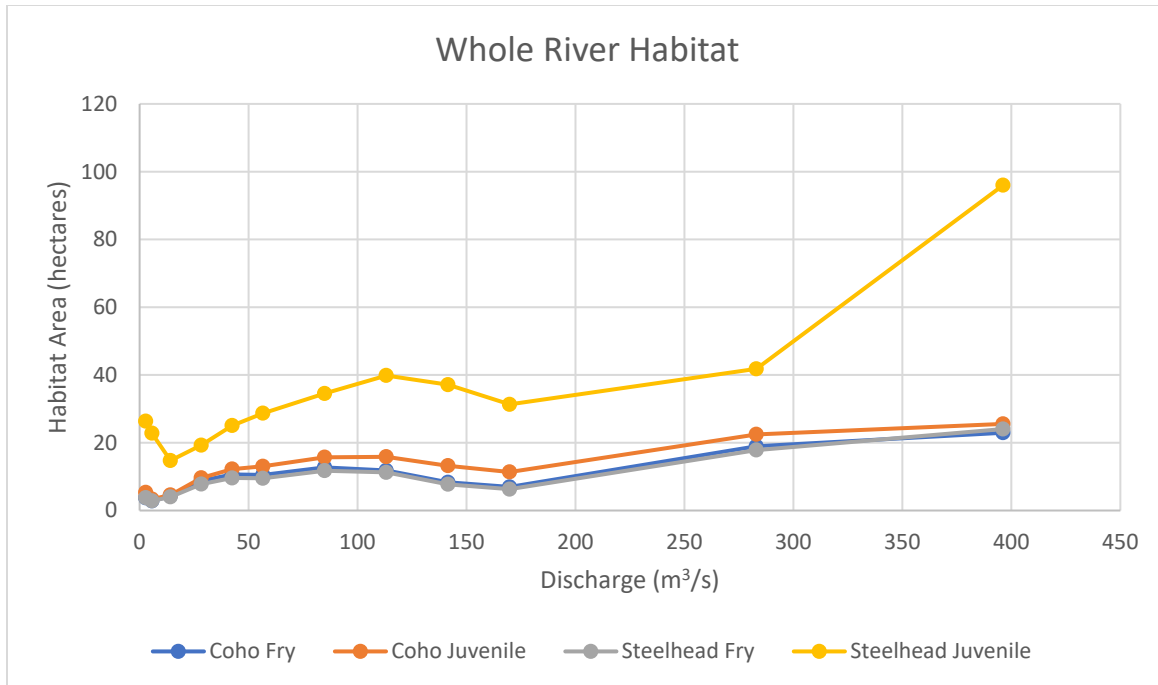
Figure 18. Discharge - Habitat Area curves for the whole river.

Figure 19 shows a comparison of how the habitat at different flow rates is spatially distributed in example restoration and non-restoration reaches. The figure shows little overlap between different flows, and also shows the prevalence of floodplain habitat, rather than habitat in the low flow channel during larger reservoir releases. In general, the restoration sites provide significantly, disproportionately more habitat than their size would be expected to generate, with ~ 15-25% (relative to the whole river) of the cumulative duration of rearing habitat availability compared to composing only ~ 6% of the simulated channel length. That substantiates their value, but a large majority of habitat for the river corridor resides outside those sites, meaning that flow releases can significantly affect overall habitat availability regardless of the sites.
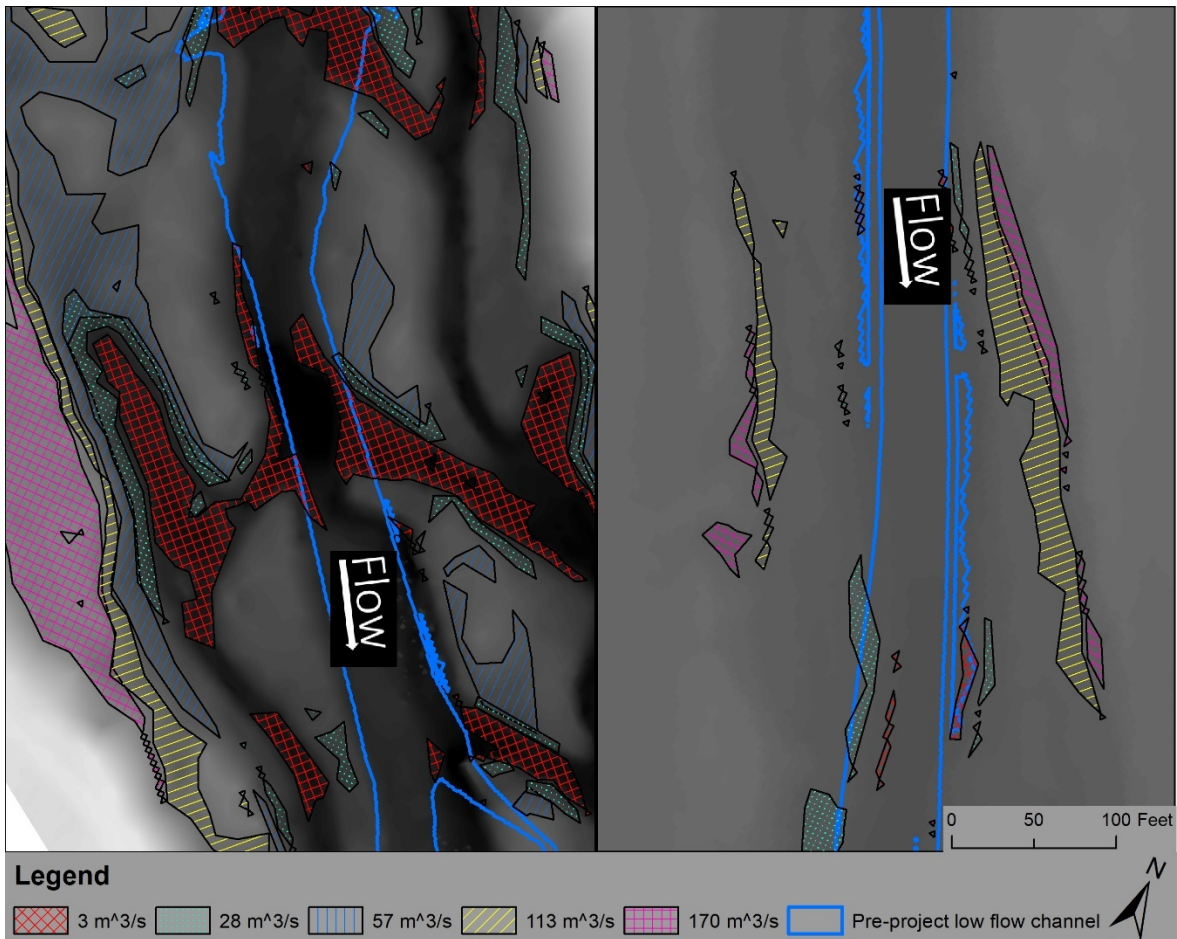
Figure 19. Spatial distribution of coho fry rearing habitat during different discharges in a restored (left) and unrestored (right) reach. Grey shading indicates topography, with darker colors indicating lower elevations.

Flow duration played a large role in determining cumulative duration of habitat availability for each scenario. For example, even though the inflow scenario had the highest peak habitat during the late February storm, this habitat was transitory in nature. To contrast, the baseflow scenario, even though it did not reach as high levels of peak habitat, had consistently large amounts of habitat for the duration of the simulation.

5.4. Release Scenario Comparison

The identified reservoir release scenarios were compared in terms of their performance in providing habitat and their effects on sediment transport, flood control and water supply (Table 5). In general, the 1k scenario was the best at meeting all objectives except rearing habitat for steelhead

juveniles. The 1k scenario had the most water remaining in the reservoir pool at the end of the season, but also had the highest peak storage (except the baseflow only scenario). Except the late February storm, which had a large peak flow in the unregulated tributaries, all scenarios met the flood control constraint of 198 m³/s (7,000 cfs) at the Geyserville gage. The 1k scenario was the only one of the operational scenarios to meet both sediment objectives relative to the baseline scenario while also leading to improvements rearing habitat availability. In terms of habitat, the 2k scenario was the only one to improve conditions for all species, life stages, and areas compared, while the 1k scenario improved conditions except in the case of steelhead juveniles. Within the restoration sites, there was an improvement over actual flows of up to 26.2 habitat hectare-days (64.95 habitat acre-days), in the case of steelhead juvenile rearing habitat provided by the 2k scenario. Figure 20 shows a comparison of total sediment out and habitat acre-days for coho fry rearing provided by the different scenarios. The 1k scenario performs the best both in terms of providing habitat and retaining sediment in the river corridor.
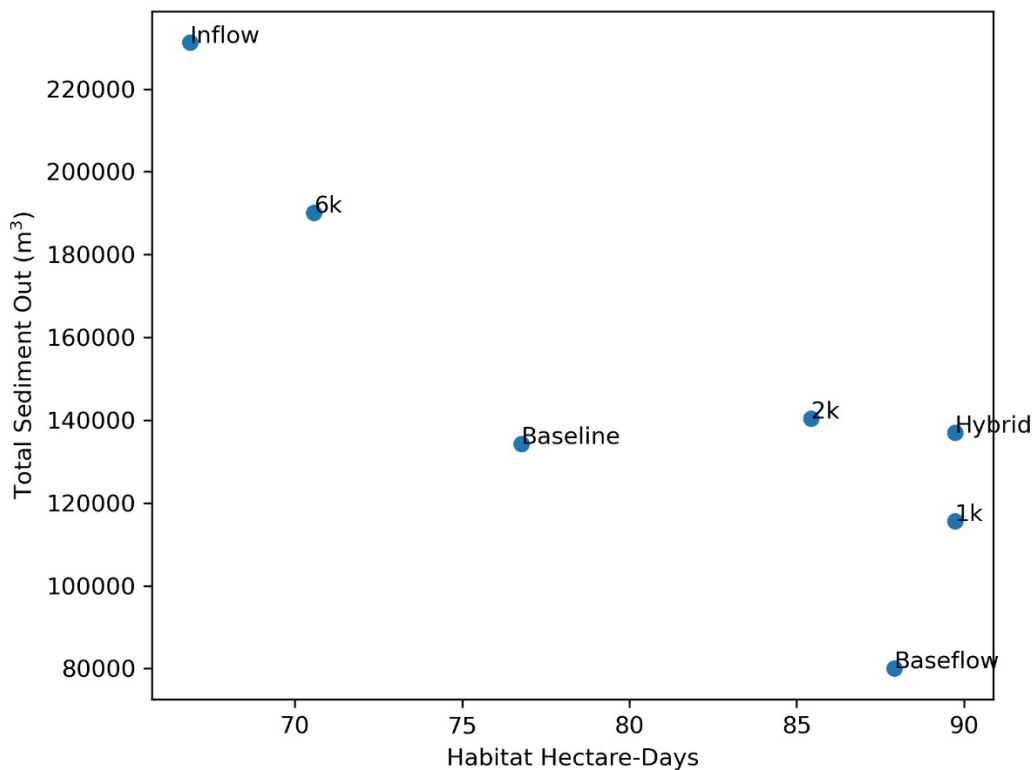
Figure 20. Comparison of habitat acre-days for coho fry in restoration sites and total sediment out among different scenarios.

Table 5. Scenario comparison summary. Bold font and shaded cell indicate best performing scenario.

Analysis	Scenario	Comparison Relative to Baseline			
		Peak Storage (Mm^3)	Final storage (Mm^3)	Date of Peak Storage (days)	Peak Flow at Geyserville (m^3/s)
Reservoir Simulation	Baseline	378	281	3/2/2019	238
	6k	**-5.6**	-17	0	**+1**
	1k	+53	**+11**	+15	+18
	hybrid	+15	-28	+15	+1
	2k	+48	-0.01	+15	+46
	inflow	-136	-39	-149	+369
	baseflow	+184	+256	+96	-8
	Scenario	Restoration Sites		Tributary Confluence Zones	
Sediment Transport	6k	+37%		-94%	
	1k	**-49%**		**+50%**	
	hybrid	+3%		+1%	
	2k	-25%		-5%	
	inflow	+46%		-178%	
	baseflow	-61%		+153%	

	Scenario	Restoration Sites				Whole River			
		cofr*	coju*	stfr*	stju*	cofr*	coju*	stfr*	stju*
Habitat Suitability (hectare-days)	Baseline	76.8	193.4	69.8	529.9	629.4	1343.1	594.4	5039.1
	6k	-6.2	-8.9	-7.2	-12.4	-141.4	-194.2	-119.1	+125.7
	1k	**+13.0**	+9.3	**+15.5**	-21.5	**+200.0**	**+141.5**	**+158.8**	-460.8
	hybrid	+13.0	+4.9	+15.5	-23.7	+200.0	+92.8	+158.8	-330.9
	2k	+8.7	**+19.2**	+6.3	**+26.3**	+22.9	+50.7	+20.6	**+145.6**
	inflow	-9.9	-38.1	-9.5	-101.8	-112.1	-140.7	-106.8	-1046.1
	baseflow	+11.2	+23.4	+8.0	+28.5	-173.6	-204.7	-129.0	+570.7

*co = coho salmon, st = steelhead, fr = fry, ju = juvenile

6. Discussion

6.1. Scenario Comparison

The comparison of alternative reservoir release scenarios found that it is possible to both increase endangered fish rearing habitat in Dry Creek and reduce deposition within the restoration sites while also having some benefit to water supply and minimal adverse impacts to flood risk management. The operational (i.e. non inflow or baseflow only) scenarios met flood risk management objectives to a similar degree as the actual dam releases, although the 1k and 2k scenarios did have higher peak flows at the Geyserville gage during the late February 2019 storm. This could likely be avoided by incorporating a flood control rule into the model, rather than just focusing on modifying the past releases as described in the methods section above. The 1k scenario had the most benefits to water supply, while others actually had reduced or the same water availability at the end of the simulation. Table 6 summarizes the relative performance of the experimental scenarios.

Table 6. Qualitative comparison of scenario performance relative to Baseline in the context of goals identified in Table 2.

Goal	Was the scenario performance better than Baseline?					
	6k	**1k**	Hybrid	2k	Inflow	Baseflow
Water supply	No	**Yes**	No	Similar	No	Yes
Flood control	Yes	**No***	Similar	No	No	No
Deposition in restoration Sites	No	**Yes**	Similar	Yes	No	Yes
Sediment retention near tributary confluences	No	**Yes**	Similar	Similar	No	Yes
Fry rearing	No	**Yes**	Yes	Yes	No	Only within restoration sites
Juvenile rearing	No, except for steelhead in whole river	**Yes, for coho only**	Yes, for coho only	Yes	No	Yes, except coho in whole river

* Based on the current flood control operation, that can be revised in the future

It is important to note that this benefit to water supply in the 1k scenario comes with an associated longer time spent in the flood control schedules of the reservoir flood management pool, which is

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something that managers try to minimize (USACE 2017). However, the simulation showed this happening in mid-March, a time of year when large atmospheric rivers with potential to rapidly increase inflows into the reservoir become less and less likely (Dettinger et al 2011). In this context, FIRO could be used to significantly reduce the risk of reservoir spill. This suggests that (1) FIRO could greatly help management of the reservoir to provide better downstream habitat and (2) the water conservation pool in the rule curve for Lake Sonoma should be raised for the spring, summer and fall months. The availability of increasingly accurate mid-range forecasts should allay reservoir managers' concerns that dam outlets do not have sufficient capacity to effectively empty the reservoir pool and prevent spill (Ralph et al 2019). As confidence in forecasts grows, reservoir release policies could be optimized across different management purposes to improve their ability to meet potentially competing goals (Alexander et al 2020).

6.2. Habitat Performance

The combination of flood release magnitude and duration led to interesting differences between cumulative habitat provided by the various scenarios. Although 57 m^3/s and 113 m^3/s (2,000 cfs and 4,000 cfs) were the maxima on the discharge-area curves for the restoration sites and whole river, respectively, they did not necessarily lead to the greatest amount of cumulative duration of habitat availability. The 1k scenario outperformed the 2k scenario in terms of fry habitat in the restoration sites, and for the whole river, the higher flood release scenarios (Baseline and 6k) were also generally outperformed by the scenarios with lower flood releases.

Because salmonid rearing is a months-long process, long duration good habitat ends up being more valuable than short duration great habitat in the context of the metrics analyzed in this study (i.e. duration of habitat availability). Although the exact implications of an increased duration of rearing habitat availability are beyond the scope of this study, there have been many studies that examine the role of seasonally accessible rearing habitats like tributaries (e.g. Ebersole et al 2006) and floodplains (e.g. Sommer et al 2001, Bellido-Leiva et al 2021). The differences in habitat quantity results between the

restoration sites and the whole river would likely be even smaller if all restoration sites were included, rather than the subset for which data were available at the time of this study. One shortcoming of the habitat modeling is that because this was done with a stable bed, changes in habitat area caused by deposition and erosion following high flows are not reflected in the model.

In the context of environmental flows, it is often recommended to return to something as close as possible to the pre-dam flow regime. However, because of the changed river corridor and sediment regime, pre-dam flows would not be ideal for meeting current management objectives of providing fry and juvenile rearing habitat in the lower watershed. This is reflected by the inflow scenario having among the lowest cumulative duration of rearing habitat availability between the different scenarios because it had short bursts of high habitat availability during storms punctuated by long periods of low habitat availability.

6.3. Sediment Transport

The 1D model results do suggest the presence of a sediment transport threshold between 28 and 57 m^3/s (1,000 and 2,000 cfs). While both 1k and 2k scenarios had less channel change within the restoration sites than the baseline releases, the 2k scenario closely followed the actual release on the longitudinal cumulative volume change curve and the 1k scenario release had more aggradation in the tributary confluence zones, suggesting a buildup of sediment within the channel, which could promote habitat development and morphological complexity in these areas where sediment is supplied (Benda et al 2004). These flow magnitudes are at least an order of magnitude below pre-dam peak flows, but the combination of narrowing/incision and reduced sediment supply have changed the morphodynamic regime.

Lane's balance, as presented above, indicates that having pre-dam outflows without the pre-dam sediment supply would lead to further incision in the river channel (Lane 1955). This conclusion is supported by the 1D sediment transport model results, which showed that the inflow release scenario led

to more erosion throughout the river corridor. This is consistent with past research showing that topographic restoration outperforms and is a necessary precursor to environmental flows in degraded, regulated rivers, especially when artificially confined and/or incised (Jacobson and Galat, 2006; Brown and Pasternack, 2008). New reservoir releases should be sized according to sediment availability to achieve current management objectives. In terms of functional flows (Escobar-Arias and Pasternack 2010, Escobar-Arias and Pasternack 2011, Yarnell et al 2015), the flow ideas presented here would be serving as a spring-recession flow, but somewhat different because they would be replacing the extended flood control dam releases. The "functional" aspect of them is that they would be providing critically needed salmonid rearing habitat while avoiding undesirable geomorphic functions of redistributing tributary-sourced sediment into restoration sites and eroding banks of Dry Creek.

6.4. Management Relative to Watershed Goals

The results from this study indicate that the changed system is such that large floods from tributaries and small floods from the dam combined with physical habitat manipulation should be sufficient to meet current habitat goals, which is similar to findings in other heavily altered systems (Anim et al 2018). In this vein, Magdaleno et al (2018) found that Dry Creek below WSD is a tributary dominated reach and the magnitudes of flow alteration and management goals are such that an outcome-based approach based on current conditions would be most appropriate for providing habitat to native biota of the watershed. These ideas have been recognized by others, that environmental flows are much more likely to be successful if they take into account the current management setting of the watershed in question (Poff 2017).

The methods used here are applicable to regulated rivers with downstream habitat restoration sites and significant tributary sediment supplies, but also could be more broadly applied in any river where wet-season reservoir releases are leading to undesired geomorphic change (or lack thereof) and habitat deterioration. As climate change is likely to lead to large peak floods in California, the way

reservoirs release these flood waters will be under increasing scrutiny and there is an opportunity to use these flood releases to support downstream habitats (Swain et al 2018, Delaney et al 2020). Combining functional reservoir releases with habitat restoration projects as shown here has been identified as a critical need for managing California's freshwater ecosystems in the coming years (Mount et al 2019).

6.5. Suggestions for Future Work

Future work would ideally integrate sediment transport and habitat suitability within a single model. Also, 2D sediment transport modeling could likely do a better job of representing morphodynamic processes within the river channel and especially in the vicinity of restoration sites and tributary confluences. In-situ measurements of bedload transport both within the mainstem of Dry Creek and in the major tributaries would greatly increase confidence in the modeling results. Also, bioverification of salmonid rearing habitat at high flows would increase confidence in the predicted amounts of habitat area (e.g. Moniz et al 2020).

6.6. Limitations

The 1D sediment transport model had numerous shortcomings that made it difficult to calibrate to the channel change data available. The deposition within some of the restoration sites is likely caused by 2D processes such as flow shadowing and differential deposition driven by gradational stratification that the model could not sufficiently characterize (Papanicolaou et al 2008). There were some issues with modeling each restoration site. For WS, the model over-predicted deposition because it showed scour of the Dry Creek channel upstream of it. It is possible that the grain size distributions in the model, due to where the data were collected, did not sufficiently characterize the armoring present in that part of the channel. For TH, the model also greatly overpredicted deposition. This could be because of the way that the cross sections in the model did not adequately characterize the flow split. FO was the one site for which the model performed relatively well. For the CH site, the major issue was the downstream

boundary condition. During large flow events on Russian River, a backwater effect occurs well into Dry Creek past where the CH site is located. This backwater effect creates a depositional environment that causes significantly more sediment to deposit beyond what would be expected in its absence. This effect was attempted to be modeled using the stage recorded at the Dry Creek mouth gage, but it caused instabilities that prevented the model from running. Because of this, normal depth was used as the downstream boundary, which made the model unable to accurately represent channel change in the lower reaches. Despite these issues, the relative effects the 1D model shows between the different scenarios are still illustrative of what could happen if the dam release rules were to change.

7. Conclusion

In the field of river restoration, the question "Restore to what?" is often asked. Sometimes, the goal is to make the site look and function like a nearby reference site, or if data are available, the goal may be to make the site look like it did pre-disturbance. However, the question is often not relevant to the situation that river managers are presented with and the goal is to have the best possible improvement from the current situation. For most dammed rivers, the best strategy from a habitat perspective will often be to remove the dams entirely, as in the Elwha River example. At the same time, removing dams completely is also not practicable in many systems like Dry Creek, as the economic benefits for urban and agricultural stakeholders are very high or irreplaceable. In these systems, reconciling the dammed river and improving degraded ecosystem conditions using functional flows presents the best approach. Coupling functional flows suited to the downstream sediment availability and habitat needs with physical habitat restoration in the river channel is critical (Whipple and Viers 2019). Taking this approach watershed by watershed has the potential to mitigate some of the harm that river development has wrought on aquatic ecosystems.

Taken together, the results from the analysis in this study suggest that having flood risk management releases in Dry Creek around 28 m^3/s (1,000 cfs) would have significant benefits for water

supply, overall river habitat development, less channel change and lower operation and maintenance costs within the restoration sites, and improve habitat conditions for a particularly sensitive lifestage of endangered salmonids in a system where they are critically limited. This comes with a cost from the risk associated with spending more time in the flood pool, but this could be ameliorated with FIRO. The long durations of late-winter and spring flood releases during wet winters can be sized appropriately to give rearing salmonids access to valuable habitat outside of the low-flow channel. The Dry Creek that exists today is a novel ecosystem and should be managed as such. The relatively reliable water source from the dam provides a valuable opportunity for providing rearing habitat for critically endangered salmonids, and it is important not to miss this opportunity by treating dam flood operation rules as unchangeable.

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