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Stream flows for salmon and society: managing water for human and ecosystem needs in Mediterranean-climate California

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Stream flows for salmon and society: managing water for human and ecosystem needs in Mediterranean-climate California

by

Theodore Evan William Grantham

A dissertation submitted in partial satisfaction of the

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University of California, Berkeley

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Professor Adina M. Merenlender, Chair
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ABSTRACT

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Professor Adina M. Merenlender, Chair

This dissertation addresses the complex relationships between human water demands and ecosystem water needs in the Mediterranean-climate region of northern California. Through a combination of long-term ecological data analysis, hydrologic modeling, and field studies of ecological-flow relationships, the research presented examines the challenges and opportunities for managing water to sustain freshwater ecosystems, with a particular emphasis on the environmental flow requirements of steelhead trout (*Oncorhynchus mykiss*). In coastal California watersheds, stream flows are increasingly impacted by small-scale withdrawals from agriculture and residential water users. While the effects of large dams have been extensively-studied, relatively little is known about the cumulative impacts of small-scale water-management practices on freshwater ecosystems. Analysis of a long-term record of fish count data revealed a strong positive relationship between stream flow and the over-summer survival of juvenile salmon and provided evidence that insufficient stream flows during the spring and summer rearing period has become an important limiting factor to threatened steelhead trout populations. These finding suggest that changes in water use practices that maintain dry season flows are critical for salmon population recovery.

A reduction in dry season water diversions could potentially be achieved by increasing local storage capacity in ponds, which could be filled during the wet season when there is greater water availability. However, increased water storage has the potential to impair winter flows, which are important for fish passage and spawning and the maintenance of habitat heterogeneity. To quantify the trade-offs between alternative water management strategies, an integrated management framework is introduced that simultaneously considers the temporal and spatial dynamics of flow regimes, and the water needs of both human and natural systems. The management framework relies on a watershed hydrologic routing model, which is useful for representing the temporal and spatial distribution of water availability and predicting the impacts of water diversions across the stream network. An examination of the potential impacts of water management practices in a Sonoma County, California watershed demonstrated that the location and size of water storage projects influences the magnitude and duration of stream flow impacts.

The findings also illustrated trade-offs between environmental flow protections and the ability for water users to meet storage demands.

Managing stream flows for ecological benefits requires an understanding of flow-habitat relationships at scales relevant to individual organisms. To assess the flow needs of adult salmon for their upstream migration, a new approach was developed for mapping stream channel topography and modeling flows in relation to fish passage suitability. The approach relies on high-resolution measurements of stream channel topography derived from terrestrial LiDAR surveys. A two-dimensional hydraulic model was then used to simulate flows in several survey reaches and identify the minimum flow required to maintain a continuous migration path of suitable passage depths for adult salmon. The modeling approach is compared to a regional formula used by State of California resource agencies to estimate salmon passage flow needs. While the results were similar, improvements to State's regional formula could be made by explicitly incorporating a variable describing channel typology.

In sum, these studies quantify the response of a threatened salmon species to flow variation, illustrate the relationships between water management and ecologically-important flow dynamics, and establish a new approach for evaluating biological flow thresholds, all of which are critical for improving the management of freshwater ecosystems in Mediterranean-climate regions.

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Chapter 1

Introduction: Managing water for human and ecosystem needs in Mediterranean-climate California

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The management of water resources is inextricably linked to the health of rivers and streams. Dams and other water management infrastructure have affected over half of the planet's largest rivers (Nilsson et al. 2005) and the overexploitation of freshwater resources continues to be a primary driver of global freshwater biodiversity loss (Dudgeon et al. 2006). Given projections of increasing human water demands and climate change impacts, pressures on freshwater ecosystems are likely to intensify (Meyer et al. 1999; Xenopoulos et al. 2005). Therefore, new approaches to water management are urgently needed to sustainably balance human and ecological needs and protect freshwater ecosystems and the services they provide.

A critical element of sustainable water management is maintaining sufficient flows in rivers to support essential ecosystem processes. This requires not only providing for minimum flows, but preserving the natural dynamic character of river flow regimes (Poff et al. 1997). Freshwater ecologists have long recognized that aquatic organisms are adapted to the natural seasonal and inter-annual variation in flows that occur in most river systems (Lytle and Poff 2004). Flow acts as a 'master variable' in river ecosystems that organizes the temporal and spatial distribution of freshwater habitats and thus has a dominant effect on community composition and species abundance (Resh et al. 1988). Because water projects and associated management practices tend to reduce flow variability (Poff et al. 2007), the restoration of natural flow regimes in river management is a necessary first step for protecting river ecosystem integrity.

Restoring natural river dynamics and establishing environmental flow protections often requires changes in water use practices (such as the operation of dams and diversions) and thus has direct, and often controversial, political and social consequences (Wohl et al. 2005). As a result, environmental water allocations have been less common in river restoration than physical habitat improvements, such as riparian revegetation, reducing sediment and reconstructing stream channels (Christian-Smith and Merenlender 2010). Furthermore, there is often substantial scientific uncertainty in predicting ecological responses to flow regimes, which has hindered implementation of environmental flow management programs (Richter et al. 2003; Arthington et al. 2006; Vaughn and Ormerod 2010). An improved understanding of ecological-flow relationships is therefore important in negotiating for environmental water allocations among competing water users and for improving the effectiveness of environmental flow restoration and protection programs (Poff and Zimmerman 2010).

Flow management is particularly important in Mediterranean climate systems because seasonal fluctuations in streamflow, as well as episodic disturbance events (e.g. inter-annual floods and drought), have a dominant influence on freshwater ecosystem structure and function (Gasith and Resh 1999). Furthermore, Mediterranean freshwater ecosystems are highly susceptible to impacts from water management operations and other anthropogenic disturbances (Alvarez-Cobelas et al. 2005). Not only does water infrastructure development tend to be extensive in Mediterranean-climate regions (Kondolf and Battala 2005), but the highly adapted nature of native aquatic species to natural flow variability may make them particularly vulnerable to activities that affect natural flow patterns (Lytle and Poff 2004).

In coastal California watersheds, streamflows that support native aquatic species are increasingly impacted by withdrawals from agricultural and residential water users (Deitch et al. 2009). The maintenance of adequate stream flows and water temperatures are particularly important to the persistence of Pacific salmon in California, which represents the southern extent of their range. Current growth in water demands, coupled with increased variability in precipitation patterns resulting from climate change suggest that streams in California will become less suitable for rearing juvenile salmonids during the dry season, which could put threatened populations further at risk of extinction. Therefore, managing water to maintain adequate flows in streams is vital for salmon population recovery and the preservation of freshwater ecosystem integrity.

In this dissertation, I explore the relationships between water management practices, natural stream flow regimes, and ecological responses in the Mediterranean-climate region of north coast California. In Chapter 2, I analyze a long-term data set of fish counts from the Russian River watershed in northern coastal California to evaluate the importance of stream flow as a potential limiting factor to threatened salmonid populations. The analysis is focused on nine years (1994-2002) of fish count data collected from four tributary streams to the Russian River in Sonoma County, California. The fish surveys were conducted in nine reaches that represent the range of stream habitat conditions occurring in the region, and were focused on the Central California Coast steelhead trout (*Oncorhynchus mykiss*), listed in 1997 as Threatened species under the federal Endangered Species Act. I employed Bayesian multiple regression models to concurrently evaluate the effects of stream flow, habitat quality, vineyard land cover, and residential development on juvenile recruitment and survival, while accounting for sampling bias resulting from repeated sampling and spatial-clustering of observations. By evaluating the same explanatory variables in separate regression models for initial recruitment and over-summer survival, our approach distinguishes the relative importance of flow, habitat, and land use factors on the early and late summer rearing phases of the salmonid life cycle and provides a quantitative description of ecological-flow relationships needed to support river ecosystem management.

The findings from Chapter 2 suggest that changes in water management that improve summer flow conditions are likely to increase juvenile salmon production. An increase in catchment storage capacity through small distributed storage ponds provides one alternative to pumping water on-demand from rivers or groundwater during the dry season. Where local water demands are met by direct surface water diversions, the ability to irrigate from stored winter water has the potential to ameliorate impacts on summer environmental flows. However, consideration must also be given to impacts on winter flows, because storage ponds are expected to reduce downstream flows until they are filled. Therefore, in a given watershed there is likely to be an optimal distribution and capacity of storage that will satisfy a proportion of human water demands while maintaining adequate stream flows to protect ecological processes.

I explore this concept in Chapter 3, which provides an extensive literature review on the effects of water management on freshwater ecosystems and presents a framework for streamflow management in Mediterranean-climate regions. The framework is based on the premise that evaluating the consequences of alterative water allocation strategies requires a spatially-explicit approach that simultaneously considers the temporal and spatial dynamics of flow regimes and the needs of both human and natural systems (Grantham et al. 2010). Previous studies that have evaluated tradeoffs between human and ecological water needs have focused on the management of individual dams on large rivers (Richter and Thomas 2007; Suen and Eheart 2006). In contrast, the proposed management framework is intended evaluate the cumulative effects of multiple, individual water users distributed throughout a watershed. The framework relies on GIS-based hydrologic model (Merenlender et al. 2008) designed to estimate discharge and water demands at all points in a stream network. The management framework is illustrated through a case study, which analyzes the effects of vineyard water use on stream flows in a small watershed in Sonoma County, California. In addition to evaluating the impacts of vineyard diversions and storage ponds, the model is used to identify potential storage sites associated with the lowest environmental flow impacts, and to examine the consequences of proposed flow management policies designed to regulate the timing of diversions to protect threatened fish populations.

The modeling framework developed Chapter 3 is designed to evaluate alternative water management strategies by quantifying the impacts of water storage diversions on flows as they are routed through a stream network. In order to assess the potential ecological effects of predicted changes in flow regimes, an understanding of local habitat-flow relationships is required. The quality and quantity of aquatic habitat available at any given time and location in the stream network is predominately controlled by stream discharge and channel properties. Thus, variation in flow regime characteristics and channel morphology along the stream network may be expected to produce spatially and temporally variable patterns in habitat suitability for fish and other aquatic organisms.

In Chapter 4, I use a 2-dimensional (2-D) hydrodynamic modeling approach to simulate flows in a natural stream channels and evaluate the relationships between discharge and hydraulic habitat parameters for salmonids. I focus on the flow requirements of migrating steelhead trout and the need to maintain sufficient water depths for successful adult fish passage. I demonstrate the utility of 2-D models in simulating water depths and flow velocities in natural, complex channels and compare the approach with alternative passage flow assessment methods commonly used in northcoast California streams. Environmental flow studies generally rely on highly simplified representations of the stream channel, either defined by regionally-based empirical relations of channel depths and drainage area or a series of transects along a stream. In contrast, the 2-D modeling approach used in this environmental-flow application is informed by a fully three-dimensional representation of the stream channel (Ghanem et al. 1996). Because the accuracy of models is linked to the quality and resolution of the channel bathymetry data (Crowder and Diplas 2000), I employed a novel survey method using a terrestrial LiDAR scanner (<0.25 m). The high-resolution survey makes it possible to specify a small cell size (0.2 m) in the model grid and simulate water depth distributions at scales relevant to individual fish. Finally, I use measured and modeled historic discharge records for the study sites to estimate the duration and frequency of passage flows and illustrate the importance of temporal variability in assessing ecological flow requirements.

In summary, information on the responses of threatened species to flow variation, the relationships between water management practices and ecologically-important flow dynamics, and the establishment of appropriate methods for evaluating biological flow thresholds are keys to improving the management of freshwater ecosystems in Mediterranean-climate and other water-stressed regions. This research is an attempt to explore these topics and to demonstrate that, despite the complexity and multiple stressors that characterize Mediterranean-climate streams, it is possible to address human water needs while ensuring the preservation of valuable and threatened ecological resources.

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Chapter 2

Relative effects of streamflow variability and land use on recruitment and summer survival of juvenile steelhead trout

Theodore E. Grantham, David A. Newburn, Michael A. McCarthy and Adina M. Merenlender

Relative effects of streamflow variability and land use on recruitment and summer survival of juvenile steelhead trout

ABSTRACT

Increasing human pressures on fresh water resources have led to global declines in freshwater fish populations and made the protection of environmental flows critical to the conservation of riverine ecosystems. Yet uncertainty in predicting ecological responses to flow alterations has hindered implementation of successful environmental flow management. An improved understanding of the relationships between stream flows and aquatic species persistence is particularly needed in semi-arid regions, where dry season stream flows are highly variable and increasingly threatened by withdrawals to meet agricultural and urban water demands. To examine the effects of summer low-flows on a threatened salmonid species, we analyzed nine years of juvenile steelhead trout (*Oncorhynchus mykiss*) count data from nine stream reaches in four coastal California watersheds. We used a Bayesian hierarchical modeling framework to examine the relative influences of stream flow, land use and habitat quality on both recruitment and over-summer survival of rearing juvenile steelhead trout. We demonstrate that over-summer survival is positively affected by the magnitude of spring flows and negatively associated with the duration of summer low flows. In contrast, higher stream flows in the spring were not associated with increased early summer recruitment. The results also suggest an adverse relationship between intensive agricultural use and juvenile steelhead trout populations, with stronger effects on the recruitment phase than on over-summer survival. These findings indicate that water quantity is a potential limiting factor to juvenile salmonid survival in coastal California watersheds, where vineyard and exurban development have expanded rapidly, and suggest that protection of summer stream flows is important for the conservation of threatened salmonid populations.

INTRODUCTION

Flow regime alterations from large dams and diversions have substantially impaired riverine ecosystems and focused attention on the importance of water management to freshwater biodiversity conservation (Bunn and Arthington 2002; Nilsson et al. 2005; Dudgeon et al. 2006). In California, water management has been a dominant factor contributing to freshwater biodiversity loss, where over 40% of native fish populations have been driven to extinction or are in serious decline (Moyle 2002). California's anadromous salmonids have been particularly impacted, with nearly all stocks currently listed as threatened or endangered under the federal Endangered Species Act (ESA). Salmon are keystone species in every stream they inhabit (Willson and Halupka 1995), are a major contributor to California's $17.3 billion dollar marine fishery, and have significant economic and cultural importance for the State's indigenous populations. Thus, the recovery of salmon populations has become a key priority for conservation planning and river restoration efforts (Roni et al. 2002; Pinsky et al. 2009).

Dam operations and water allocation policies are increasingly being revised reduce the adverse ecological effects of water management and protect instream flows required for fish and other aquatic species. Several innovative approaches have been developed to define instream flow standards for freshwater ecosystem maintenance (Richter et al. 1997; King and Brown 2006; Poff et al. 2010), yet uncertainty about ecological flow needs together with conflicting human water demands has hindered implementation of environmental flow management (Arthington et al. 2006). The identification of quantitative ecological-flow relationships is complicated because the abundance and composition of aquatic communities fluctuate in space and time in response to multiple, interacting environmental factors, which makes it difficult to isolate the functional role of stream flow. Long-term data linking biological and physical conditions are often required to accurately model hydrological variability in relation to aquatic species populations, yet most studies are conducted at scales that do not fully account for the range of temporal variability and spatial heterogeneity of stream ecosystems (Webb et al. 2010). Thus, there is a persistent need for data collection efforts that make it possible to infer quantitative relationships between flow variability and ecological response (Poff and Zimmerman 2010).

The importance of dry season stream flows to the persistence of fish populations has received increased attention in recent years (Magoulick and Kobza 2003; Bond et al. 2008; Perry and Bond 2009; Riley et al. 2009)), yet surprisingly little is known about the role of low flows in supporting threatened and endangered fishes in the western United States. The dry season is the juvenile rearing period for many native fish species including salmonids and coincides with peak water demands for agricultural and urban uses. As in many Mediterranean-climate regions of the world (Underwood et al. 2009), coastal California watersheds are experiencing extensive land use change associated with the expansion of vineyards and exurban development (Merenlender et al. 2005; Newburn and Berck 2006). Because vineyards in smaller, upland watersheds largely rely on surface water diversions for their irrigation requirements (Deitch et al. 2009), there is direct competition over water needed for agricultural production and the maintenance of freshwater habitat. Low-flows critical for sustaining rearing habitat for juvenile salmonids are likely vulnerable to impacts from water withdrawals, which could put threatened populations further at risk of extinction. Thus, understanding the relationships between stream flow, land use, and salmonids is essential to proactively manage and define conservation policies for these endangered freshwater species.

In this study, we analyze a unique long-term biological data set to evaluate the role of dry season flows as a potentially limiting factor to juvenile salmonids, and assess other local and landscape factors that influence their abundance and survival. The analysis focuses on nine years of fish count data collected from nine stream reaches in four watersheds to estimate summer recruitment and survival of Central California Coast steelhead trout (*Oncorhynchus mykiss*), listed in 1997 as threatened species under the federal ESA. We employ a Bayesian hierarchical modeling approach (Gelman and Hill 2007) because it offers a flexible framework to analyze data from multiple non-replicate sampling units and has been shown to greatly improve inferential strength in data-poor situations common in ecological monitoring (Webb et al. 2010). Specifically, Bayesian regression models with non-informative priors were used to concurrently evaluate the relative effects of stream flow, habitat quality, vineyard land use, and residential development on recruitment and survival, while accounting for potential bias resulting from repeated sampling and spatial-clustering of observations. We use a Poisson regression model to analyze the early summer recruitment and a logistic regression model to analyze over-summer survival. Because the regression models include variables that primarily operate at different spatial and temporal scales, their relative influence on recruitment or survival can be more accurately estimated.

Habitat units in the same reach, for example, may have different habitat quality but experience the same amount of upstream land use (e.g. vineyard), while stream flow primarily varies temporally in response to precipitation differences between years. By evaluating the same explanatory variables in separate regression models for initial recruitment and over-summer survival, our approach distinguishes the relative importance of summer flows, habitat, and land use on the early and late summer rearing phases of the salmonid life cycle. This is critical for guiding the management of freshwater resources and developing effective conservation strategies for threatened salmonid populations.

METHODS

Study Area

Maacama Creek, Mark West Creek, Santa Rosa Creek, and Green Valley Creek are tributaries of the Russian River in northern coastal California (Figure 2.1). These medium-sized drainages (12–128 km^2) are located in the south-eastern portion of the Russian River basin (3,850 km^2), with elevations ranging from 15 – 40 m above sea level at their confluence to 400 m in their headwaters. The study area is located in the coastal Mediterranean-climate region, characterized by cool, wet winters and hot, dry summers. Virtually all annual precipitation (average of 90 cm) occurs as rainfall between November and March, and as a result, stream flows peak in the winter months and gradually recede through the spring to approach or reach intermittency by the end of the dry summer season. Mean dry season (defined as May 1–Sept. 30) discharges in the larger study basins (>100 km^2) typically range from 0.03 to 0.07 $m^3 \cdot s^{-1}$, while dry season discharges in the smaller study basins (<100 km^2) are typical less than 0.02 $m^3 \cdot s^{-1}$. Mixed-hardwood forests, oak savannas and grasslands constitute the majority of the natural vegetation cover of the study basins. The region is known for its high quality wine grape production, and vineyards can be found in valleys and on moderately-sloped hills. Santa Rosa (population ca. 160,000) and Healdsburg/Windsor (population ca. 30,000) are the major urban centers in the Russian River basin. However, rural residential and vineyard development occupy more land area and represent the most rapidly expanding land-use types in the region (Merenlender et al. 2005; Newburn and Berck 2006).

Juvenile steelhead surveys

Fish sampling was conducted between 1994 and 2002 by fisheries biologists as part of a long-term environmental assessment program funded by the City of Santa Rosa (Merritt Smith Consulting 2003). The surveys were focused in three sampling reaches (lower, middle, and upper) of Mark West and Maacama Creeks, two reaches (middle and upper) of Santa Rosa Creek, and one reach in Green Valley Creek (Figure 2.1). Reaches located within the same stream were separated by at least 4 km. Between 6 and 10 physically discrete habitat units (e.g., pool or riffle) were selected for sampling in each reach. Habitat unit dimensions were measured once in July 1994 and, on average, were 4.3±1.9 m (mean±SD) wide and 27.0±23.9 m (mean±SD) long, with mean maximum depths of 0.7±0.3 m (mean±SD). The selection of habitat units was biased toward deep (>0.5 m) pools that provided the best habitat in the reach because most fish tended to concentrate in those units during the dry season. In particularly dry years, deep and shadier pools offer the only suitable habitat available in the summer months, though they represent a small proportion of the stream length. The same habitats units were sampled in consecutive years but occasionally winter storms would modify or eliminate particular pools. In those instances, a new unit in the same stream reach would be sampled the following year. Fish counts were conducted semi-annually in the beginning of summer (June–July) when flows between pools approached intermittency, and again in early fall (October) prior to the onset of the rainy season (Figure 2.2). We define the initial summer count as "recruitment," noting that the observed abundance does not reflect total juvenile production given that young-of-the-year fish emerge in the spring and some may have died or moved out of the reach prior to sampling. The summer flow recession is associated with decreasing surface water connectivity, which confines to fish to remaining wetted channel habitats and prevents the movement of juvenile steelhead to and from adjacent habitat units between the sampling events. However, because the movement of individual fish was not monitored, we consider the change in fish abundance over the summer to be an estimate of "apparent" survival. In total, 54 and 59 habitat units were sampled each year in the early and late summer, yielding a sample size of 500 paired observations over the period of the study.

All field sampling was conducted by the same 2-person team throughout the 9-year period (Merritt Smith Consulting 2003). Fish were collected in each habitat unit by repeated passes through the stream with pole seines, approximately 1.2 m deep by 1.5, 4.5 or 7.5 m long, with a 0.5 cm mesh size. Prior to sampling, blocking nets were placed across the ends of each habitat unit and any mobile instream objects that could obstruct the nets were temporarily removed. Multiple passes were made with the seines until no individuals were captured relative to the numbers captured in earlier passes. Typically, three to five passes were made but in structurally complex units up to ten passes were necessary to exhaustively sample a unit. After each pass with the seine, captured fish were temporarily relocated to aerated buckets, then sorted by species and counted.

Stream flow variables

To estimate flow conditions at each of the reaches during the nine years of fish sampling, we developed a rainfall-runoff regression model that predicts mean daily flows ($m^3 \cdot s^{-1}$) based on daily rainfall records. We used U.S. Geological Survey flow data from Maacama Creek (USGS Station # 11463900; 1961–1980) and Santa Rosa Creek (USGS Station # 11465800; 1959–1970), and rainfall records from a nearby precipitation station (Healdsburg, CA, USA) to fit a log-linear regression model of mean daily flows to rainfall occurring in antecedent periods (e.g., prior 1–7 days, 8–15 days, 16–30 days, etc.). We then used precipitation records from the study period (1994–2002) to estimate flow at each of our study sites. Flow measurements taken at each reach after the study period (2003–2005) were used to calibrate the model by rescaling predicted flows by a constant that minimized the residuals between observed and predicted discharge values. Because we aimed to evaluate the effects of inter-annual stream flow variability on salmonid populations at multiple locations, we needed to account for variation in discharge associated with the differences in catchment area among the study sites. We therefore normalized discharge by the catchment area to develop hydrologic metrics for our statistical model that described hydrologic conditions independent of catchment size

We focused our analyses on two hydrologic metrics that describe variation in inter-annual and seasonal flow patterns. Median spring (1 March–30 June) flow quantifies the magnitude of daily stream flows during the early rearing period for juvenile salmonids (Figure 2.2). Furthermore, median spring stream flow is highly correlated with summer flows and therefore provides a metric of stream conditions throughout the dry season period. The second hydrologic variable evaluated was summer low-flow days, defined as the duration in which daily flows fell below the lower quartile of values observed in each reach over the nine-year study period (1 July–30 September). The dry season is characterized by a gradual and consistent flow recession and this low-flow variable captures how quickly flows receded each summer relative to reach-specific hydrologic conditions observed over the period of record (Figure 2.2). The two hydrologic variables therefore represent both the magnitude of flows and the timing in which streams approach ‘low-flow’ conditions for each reach of each year of the study. The median spring flow variable reflects stream flow conditions at the beginning of the dry season (the initial sampling) and the low-flow variable measures the duration in which fish experience severe low-flow conditions through the end of the summer (the final sampling). Other hydrologic metrics were evaluated, but not included in the model because of their high correlation ($r>0.5$) with the other stream flow variables. During the study period, median spring daily flow varied from 0.01–0.03 $m^3 \cdot s^{-1}$ among all sites in the lowest runoff year to 0.05–0.16 $m^3 \cdot s^{-1}$ among all sites in the highest runoff year. Over the study period, the number of low-flow days ranged from 0–83 days.

Land use and habitat variables

In 1994, a habitat assessment was conducted by the fish-sampling team in all study reaches. The dimensions of each sampled unit were measured and a qualitative habitat suitability score was assigned based on the presence of preferred features of rearing juvenile steelhead at least one-month post-emergence (fork lengths >30 mm). Habitat units were assigned a score from 1 to 3 (with 3 representing the highest habitat suitability) based on the extent of instream shelter, riparian vegetation cover (and/or shade), and deep-water areas. Due to the nature of the assessment method, the potential effects of different habitat components on fish abundance and survival could not be evaluated independently. Because the number of units with a rating of 3 ($n = 388$) was significantly greater than units with ratings of 1 ($n = 63$) and 2 ($n = 49$), for regression purposes we transformed the habitat rating into a dichotomous variable representing high (scores of 3) and low (scores of 1 and 2) habitat suitability. Study reaches in smaller, upper watersheds tended to have a higher proportion of units with high habitat suitability ratings than those in the larger watersheds, although units with high-suitability ratings occurred in all of the reaches.

To quantify the effects of land use on summer fish densities, we measured vineyard use and road density (as a proxy for exurban development) in the watershed above each study reach. We selected these variables to reflect the dominant land uses in the study watersheds that have previously been shown to influence stream flow and habitat quality (Lohse et al. 2008; Deitch et al. 2009). Vineyard use was measured within a Geographic Information System (GIS) based on aerial photographs taken in 2006. The total vineyard area was divided by the drainage area above each study reach to obtain percent vineyard use, which ranged from 1.18% –6.85%. We also calculated road densities (km road per km^2) within a GIS based on 1:24,000 scale USDA Forest Service Cartographic Feature Files, published in 2002. Road densities ranged from 0.45–1.6 $km \cdot km^{-2}$ in the drainages areas above each of the study reaches.

Modeling Recruitment and Survival

We developed separate regression models for recruitment and over-summer survival to evaluate the relative effects of stream flow variability, habitat, and land use on the different juvenile rearing periods. Because the explanatory variables vary at different spatial and temporal scales, their relative influence on recruitment and survival could be assessed in a multiple regression modeling framework. For example, median spring flow and low-flow days primarily vary between years, while the land use and habitat variables vary between watersheds and were relatively constant over the sampling period. Vineyard use and road density are calculated for the drainage area above each reach, and therefore all sampled pools within a given reach experience the same effects of upland land use. In contrast, the habitat variability is measured at each pool and thus measures variation in habitat quality between pools within the same reach.

For recruitment (n = 500), we modeled the unit-specific fish abundance, N, as Poisson-distributed with parameter λ

$$N_{ijt}|\lambda_i \sim \text{Poisson}(\lambda_{ijt}) \qquad [1]$$

where N_{ijk} is the summer count at unit i in reach j and year t. The logarithm of the expected abundance (λ_{ijt}) was then analyzed in a Poisson-regression formulation with reach-specific random effects to model systematic variation in recruitment among sampled units as a function of explanatory variables

$$\log(\lambda_{ijt}) = \log(L_i) + \beta_0 + \beta_1 Q_{jt} + \beta_3 H_{ij} + \beta_4 R_j + \beta_5 V_j + b_j + \kappa_{ijt} \qquad [2]$$

where L_i is an exposure variable to account for difference in unit lengths, β are the model coefficients for the mean intercept (β_0), median spring runoff (Q_{jt}), habitat suitability (H_{ij}) road density (R_j), and vineyard use (V_j). Habitat units were sampled within each of the nine sampling reaches, so we included the reach-specific random effect, b_j, to account for the influence of unobserved covariates within each reach. The random-effects specification assumes that the effect of the reach environment is common to all habitat units sampled in a given reach. Hence, the model accounts for error correlation over time and yields robust standard error estimates of the regression coefficients. An additional error term, κ_{ijt} ,was included to account for overdispersion (variance of the counts exceeds the mean of the counts) in the data (McCullagh and Nelder 1989). Fish counts are commonly scaled by the area or volume to control for differences in the sizes of the sampled habitat unit. We chose to scale the counts by length because habitat unit dimensions were not measured at each sampling event and the wetted widths of individual habitat units are likely to fluctuate in response to changes in flow. In contrast, the lengths of habitat units are relatively insensitive to flow changes and therefore are a more appropriate scaling metric for these data.

In a separate analysis, we used a hierarchical modeling framework (Gelman and Hill 2007) to identify the effects of stream flow variability, habitat, and land use on apparent over-summer survival. We estimated survival probability, p, of fish within each habitat unit using a binomial distribution of the number of surviving fish, S, sampled from those counted in the early summer survey, N

$$S_{ijt} \sim \text{Binomial}(N_{ijt}, p_{ijt}) \qquad [3]$$

where p_{ijt} is the estimated survival of individuals in habitat unit i in reach j and year t. We assume that the population in each unit is closed to immigration and emigration between sampling events and that counts on the same unit in different years are independent. We also excluded initial fish counts with no fish present (n = 63) to yield 437 paired observations. We then employed the logit link function to express survival probability for the unit as a linear function of the explanatory variables

$$\text{logit}(p_{ijt}) = \beta_0 + \beta_1 Q_{jt} + \beta_2 L_{jt} + \beta_3 H_{ij} + \beta_4 R_j + \beta_5 V_j + \beta_6 D_{ijt} + b_j \quad [4]$$

where β are model coefficients for the explanatory variables and b_j is a reach-specific random effect term. The same covariates used in the recruitment model were included in the survival model [median spring runoff (Q_{jt}), habitat suitability (H_{ij}), road density (R_j), and vineyard use (V_j)] in addition to the number of summer-low flow days (L_{jt}). We also included initial fish density [fish per length of the sampled unit (D_{ijt})] in the model to control for potential density-dependent effects on survival. Correlation among the explanatory variables in both models were weak ($r<0.4$).

Bayesian Regression

The analysis was done in a Bayesian framework because it is well-suited to fit hierarchical models that accommodate variation in the data due to overdispersion and/or non-independence of observations (Congdon 2006). We conducted model fitting with Monte-Carlo Markov chain (MCMC) methods (Gilks et al. 1996) using the software package WinBUGS (Spiegelhalter et al. 2003). Prior distributions for regression coefficients were normal with means of 0 and standard deviations of 1000. The random effects were assumed to be drawn from a normal distribution with mean zero and a standard deviation that was estimated from the data. Priors for the standard deviations of the random effects were uniform between 0 and 100. The vague priors specified in the model meant they had little influence on the posterior distributions of the parameters, resulting in a similar shape to the likelihood function. For each model, we simulated three MCMC chains with different initial values for 200 000 iterations after a burn-in of 50 000 iterations, thinning by a factor of 10 to reduce autocorrelation in the sample. The burn-in was more than sufficient to ensure the MCMC samples were being drawn from the stationary distribution, based on visual inspection of chain convergence and Gelman-Rubin diagnostics (Gelman and Rubin 1992). To improve the efficiency of the MCMC sampling, all explanatory variables were centered by subtracting the mean. The focus of our analysis is on the resulting posterior distributions of parameters of the recruitment and survival models. For each parameter, we report the mean and 2.5th and 97.5th percentiles of the posterior distribution. This interval represents a 95% Bayesian credible interval (CI), which expresses the level of uncertainty in the parameter estimate. If working within the null-hypothesis testing framework, a variable effect with a 95% CI that does not encompass 0 is similar to rejecting the null hypothesis of no effect. For the recruitment model, the relative importance of each variable was assessed by calculating their multiplicative effect, given by the exponent of the standardized coefficient

$$E_k = \exp(\beta_k \times range_k) \quad [5]$$

where E_k is the multiplicative effect of variable X_k, β_k is its regression coefficient and $range_k$ is the range of the variable observed during the study. For the survival model, the multiplicative effect of each explanatory variable was calculated as the ratio of predicted survival probabilities

$$E_k = \frac{p_{\max}}{p_{\min}} = \frac{1+e^{-(\bar{\beta}_0+\beta_k\times\min_k+\bar{\beta}_1\times\bar{X}_1+\ldots+\bar{\beta}_5\times\bar{X}_5)}}{1+e^{-(\bar{\beta}_0+\beta_k\times\max_k+\bar{\beta}_1\times\bar{X}_1+\ldots+\bar{\beta}_5\times\bar{X}_5)}} \quad [6]$$

where E_k is the multiplicative effect and $p_{\max}$ and $p_{\min}$ are survival estimates evaluated at the maximum and minimum data values for each of the k explanatory variables (X) over the 95% CI of the parameter estimate (β_k), while holding all other variables constant at their mean value. A multiplicative effect less than one predicts a decrease in the response variable, while an effect greater than one predicts an increase. An explanatory variable with a multiplicative effect substantially different from one is likely to have a biologically important effect. The 95% credible interval shows the range of plausible values for the multiplicative effect of a variable, with an effect of one indicating no change in recruitment or survival probability (i.e., no detectable effect) over the range of the variable.

RESULTS

Seasonal and annual variation in fish counts

Over 90% of individual salmonids captured during sampling were young-of-the year (age 0+) steelhead trout and few older-age class individuals were present. The mean density of juvenile steelhead trout in the early summer was 1.5±2.5 individuals per meter stream length (mean±SD ind·m^{-1}), and ranged from 0–27 ind·m^{-1} (min–max) based on 500 habitat-unit observations over the study period. Mean fish densities in the early summer did not substantially differ among years and consistently fell between 0.9 and 2.0 ind·m^{-1} (Figure 2.3A). However, early summer fish densities were markedly different between reaches (Figure 2.3B). Middle and upper stream reaches with smaller drainage areas (12–60 km^2) generally supported higher densities than lower stream reaches (drainage areas 100–130 km^2). In the early summer, mean fish densities in the upper reaches ranged between 0.8 and 3.7 ind·m^{-1}, while mean fish densities in the three lower reaches were between 0.3 and 0.5 ind·m^{-1}.

Between the early and late summer counts fish densities declined, on average, by 1.0 ind·m^{-1} (or 66%). Variation in fish densities also decreased from the early summer, with late summer densities ranging from 0–11 ind·m^{-1} (min–max). Changes in fish densities within a year did not appear to be influenced by initial summer density. That is, the years with the highest initial fish densities were not necessarily those with the highest late summer fish densities (Figure 2.3A). However, differences in fish densities between reaches often did persist to the late summer, such that reaches with higher fish densities in the early summer also tended to have higher fish densities in the late summer (Figure 2.3B).

Fish recruitment

The Poisson regression model for early summer recruitment indicated that vineyard and habitat variables were significantly related to recruitment patterns. There was a positive association of local habitat quality on early summer fish density and a negative association with vineyard use (Table 2.1). Juvenile recruitment was predicted to be 1.54 [95% credible interval (CI): 1.02, 2.28] times greater in units with high habitat quality than in units with poor habitat quality (Figure 2.4). Vineyard had a strong negative multiplicative effect of 0.12 [95% credible interval (CI): 0.02, 0.56], which correspond to the prediction that early summer recruitment densities are eight-fold higher in watersheds with the lowest level of vineyard development compared with the highest levels of vineyard development. Road density had a non-significant multiplicative effect of 0.97 [CI: 0.19, 5.33]. The regression coefficient mean for median spring runoff was positive. However, the 95% CI of the coefficient effect was [0.83, 3.01] and encompassed one, suggesting variation in spring stream flow levels do not explain patterns in recruitment (Table 2.1).

Fish over-summer survival

Survival estimates of juvenile steelhead from early to late summer ranged from 0.5% to 84%, with a mean of 29±17% (mean ± SD) across all sampled units. The logistic regression model indicated that both median spring stream flow and summer low-flow days were important factors in explaining variation in over-summer survival (Table 2.2). The model predicted an increase in survival by a factor of 3.79 [CI: 3.71, 3.86] when comparing years of lowest and highest median spring stream flows (Figure 2.4). Apparent survival showed an approximately linear, increasing trend across the range of observed median spring flows (Figure 2.5A), but exhibited a downward trend with increasing low-flow days (Figure 2.5B). The decline in apparent over-summer survival with increasing low-flow days corresponds to a multiplicative effect of 0.84 [CI: 0.77, 0.92], which indicates that habitat units in the driest years are expected to have 15% lower survival than observed in wetter years with the fewest number of low-flow days (Figure 2.4). As with recruitment, spatial variation in habitat quality within reaches does explain some variation in over-summer survival. The model predicts that units with high habitat quality will, on average, have survival rates 1.19 [CI: 1.07, 1.31] times higher than units with low habitat quality (Figure 2.4). Spatial variation in road density above each sampling reach did not explain observed variation in apparent over-summer survival, while vineyard use appeared to have a negative effect [CI: 0.10, 1.06].

DISCUSSION

Our results indicate that stream flow, habitat and land use variables are significant in explaining patterns in juvenile steelhead trout abundance, but importantly these factors have different effects on the recruitment and over-summer rearing stages of threatened juvenile steelhead trout populations (Figure 2.4). First, higher stream flows are positively associated with over-summer survival but the magnitude of median spring flows does not appear to have a significant influence on early summer recruitment. Second, habitat suitability has a positive association with both recruitment and over-summer survival. Third, vineyard land use had negative effects on both stages, but the effect was more significant on the initial recruitment stage. Lastly, road density did not explain patterns in recruitment and survival.

Effects of stream flow variability

The low apparent over-summer survival rates suggest that the dry season is a significant period of stress for rearing juvenile steelhead trout in the study region. A mean survival of 29% (median of 25%) is substantially lower than rates reported from other Pacific coast streams. For example, Boss and Richardson (2002) found that summer survival of cutthroat trout in two coastal streams in British Columbia, Canada ranged from 40 to 100%, but had a median of 95%. Studies conducted by Harvey and others (2005, 2006) in the temperate-climate region of northern California also indicated that the over-summer survival of rainbow trout in coastal streams was generally greater than 60%. Stream flows appear to be an important mediator of rearing habitat conditions in the dry season. While it is sensible to expect decreased juvenile fish survival during severe drought years, the consistent increase in survival over the range of observed flow conditions indicate that relatively small changes in flows can affect juvenile salmonid survival rates (Figure 2.5).

There are a variety of plausible mechanisms by which increased dry season flows could improve the fitness and survival probability of juvenile steelhead. For example, flows regulate the input of invertebrate drift, the primary food source of trout and other salmonids, and thus increases in flows could be expected to increase food availability and improve the fitness and survival of fish during the dry season (Harvey et al. 2006; Hayes et al. 2008). Flow patterns are also tightly coupled with stream thermal regimes and other water quality parameters and the severity and duration of adverse water quality conditions are likely to be increased under low flow conditions. For example, high air temperatures and low flows in the summer can elevate stream temperatures above critical thermal maxima for salmonids (Myrick and Cech 2004), concentrate pollutants to toxic levels, and decrease dissolved oxygen concentrations (Nilsson and Renöfält 2008). Finally, flows directly control the velocity, depth and volume of water in the stream channel and thus mediate the size and suitability of habitat (Dewson et al. 2007). Therefore, higher low flows are likely to increase the size and suitability of fish habitat by maintaining riffle connectivity and pool depths, potentially reducing the risk of mortality by predation, competition, and stranding. Despite the strong association of flows with apparent over-summer survival, interannual variability in spring flows did not appear to influence the early-summer abundance of juvenile steelhead. This contrasts with findings by Lobón-Cerviá (2007), who found that discharge in March explained substantial amounts of spatiotemporal variation in recruitment of resident brown trout. The fact that early summer densities varied little between years within reaches suggests that, during late spring and early summer, juvenile steelhead populations in the study streams are controlled by factors other than flow.

Effects of habitat and land use

Our results suggest that vineyard use has a significant effect on both recruitment and survival. The negative association of vineyard use on juvenile salmonids is likely related to the impacts that intensive agriculture has on both habitat and stream flows. The direct effects of land-use conversion is consistent with previous studies that document impacts to salmonid habitat and populations from the conversion of wild lands to agriculture, managed forest, urban, and rural residential land uses (e.g., Paulsen and Fisher 2001; Bilby and Mollot 2008; Lohse et al. 2008). The magnitude of the effect of vineyard on recruitment is notable, with a predicted eight-fold decrease in early season abundance when comparing reaches of the lowest and highest levels of vineyard cover, conditional on all other variables held at their mean value. Vineyards often rely on groundwater pumping or direct surface water abstraction to meet their water demands, which have been shown to impair stream flows in the late spring for frost protection and summer for heat protection (Deitch et al. 2009). Thus, vineyard water use may directly impact juvenile salmonids by dewatering streams, reducing habitat availability and causing mortality from stranding. Vineyard and exurban development in the region are also associated with increased fine sediments inputs to streams (Lohse et al. 2008), and thus may be indirectly impacting salmonids through habitat degradation. The rapid expansion of vineyard and exurban development in the study area suggests that the adverse effects of surrounding land use on freshwater ecosystems are likely to increase in severity and warrant greater attention.

Vineyard and road density are variables that overlap in geographic extent for reaches nested in the same watershed, which introduces spatial autocorrelation within the data. Furthermore, the position of a reach within the stream network is likely correlated with spatial trends in habitat and watershed attributes, which could confound the apparent influence of landscape variables. This is a common problem for watershed scale studies and can make it difficult to interpret the results of regression models (King et al. 2005). In this analysis, we reduce the potential influence of reach position by sampling from four separate watersheds, limit our analysis to land use variables that are weakly correlated, and evaluate the effects of land use for exploring potential causal relationships and not for predictive purposes. Furthermore, the habitat variable used in the model reflects some of the between-reach differences in habitat quality associated with watershed size. Finally, the incorporation of random effects in the model accommodates the effects of unobserved covariates operating at the reach scale that may also be related to nested, spatial structure of the data.

The strong positive association of habitat quality on rearing steelhead trout that we observed is entirely consistent with previous studies documenting the important role of habitat features on juvenile salmonid growth and survival (e.g., Lonzarich and Quinn 1995; Harvey et al. 2005; Johnson et al. 2005). Within all of the study reaches, sampled units with a high habitat suitability rating had significantly higher initial fish densities and over-summer survival than units with low habitat quality ratings. Notably, the lack of water temperature data and detailed information on physical habitat conditions during the study period makes it difficult to ascertain how different habitat components (e.g., riparian cover, instream shelter, embeddedness, and depths) affect the abundance and survival of juvenile steelhead. Furthermore, because physical habitat information was not recorded during each year of sampling, interactions between flow patterns and habitat features over time cannot be investigated. Nevertheless, the model makes efficient use of the available data by accounting for the important influence of local habitat features on fish recruitment and survival relative to other relevant environmental factors.

These data offer a unique opportunity to evaluate how the influence of a similar set of variables on juvenile fish populations can vary over the course of the dry season. Juvenile salmonid counts are typically conducted in the fall and monitoring inter-annual changes in fish densities is considered important for predicting future trends in adult populations. However, because fall densities reflect an aggregated measure of spring recruitment, movement, and mortality, it is difficult to identify the timing and manner in which different natural and anthropogenic factors affect the abundance and distribution of juvenile fish. In this study, we take advantage of variables operating at different spatial and temporal scales to identify the relative influence of stream flow, land use, and habitat quality on initial recruitment and over-summer survival. Hence, vineyard use operating at the watershed level and habitat quality operating locally at the pool level are both significant factors that influence the early summer fish densities. Stream flow does not appear to have a significant effect on recruitment. Rather, the temporal variation in stream flow between wet and dry years has the largest mediating impact on the over-summer survival phase.

Our findings underscore the importance of long-term survey data for assessing ecological responses to environmental and anthropogenic change. In climatic regions marked by high interannual variability and dynamic flow regimes, long term data are important for distinguishing the effects of multiple natural and anthropogenic stressors on freshwater ecosystems (Osenberg 1994; Bêche et al. 2009). Many studies have attempted to correlate organism performance with environmental factors but few have proven useful in guiding management decisions because links between the two are often obscured by many factors. Identifying environmental controls on anadromous fish populations is particularly difficult because of their complex life histories and exposure to multiple stressors in both marine and freshwater environments. We are able to perform these analyses because data were available from the range of dry, wet, and normal rainfall years over the nine-year study period. Furthermore, because the sampling was stratified by both space and time, it was possible to distinguish the relative effects of stream flow conditions, land use development, and habitat quality occurring at different spatial and temporal scales. The long-term data necessary to investigate ecological-hydrological relationships are rare and is a major obstacle to effective freshwater ecosystem conservation and river-management (Vaughan and Ormerod 2010). It is also difficult to obtain all of the site and landscape variables that may influence salmon recruitment and survival in upland tributaries. Consequently, the strength of model effects of may change with the incorporation of additional explanatory variables that are currently unavailable. Additional research is needed to quantify the potential effects of other variables and achieve a mechanistic understanding of the factors that influence juvenile recruitment and survival. Nevertheless, opportunistic analyses of available data is an important first step for detecting and quantifying relationships between environmental variability and the spatial and temporal patterning of juvenile salmon populations.

Conclusions and management implications

Given the widespread decline of anadromous fish species, there is a pressing need to implement more effective regulatory measures to protect freshwater habitats. Yet there remains substantial uncertainty on the flow requirements of aquatic species and how they will respond to flow regime alterations, which makes it challenging to formulate management recommendations. For streams in semi-arid regions such as Mediterranean-climate California, the abundance and survival of juvenile salmonids during the summer dry season are linked to stream flow regimes and land use practices. Our evaluation of the relative effects of key environmental variables on different juvenile rearing periods can help direct future investigations and inform management interventions. For example, vineyard use had a strong negative association with recruitment, which suggests that future research on the ecological effects of agricultural land use conversion and management should focus on potential alterations to habitat conditions in the spring and early summer. In contrast, the flow variables were important for predicting over-summer survival but did not have as great an influence on spring recruitment, indicating that flow recovery efforts to benefit fish should focus on the low-flow summer period. Our findings suggest that changes in water management that improve summer flow conditions are likely to increase production of juvenile salmonids, which demographic modeling (Kareiva et al. 2000) and empirical studies (Mathews and Olson 1980) have shown to relate well to adult returns. Water use impacts on summer flows could potentially be achieved by changing agricultural water use practices. For example, vineyard landowners could potentially increase onsite storage and capture high winter flows to meet their water demands during the summer growing season (Grantham et al. 2010). Projected climate change and population growth in Mediterranean-climate and other semi-arid regions will unquestionably increase pressures on water resources and intensify impacts to freshwater ecosystems already in severe decline. Therefore, the identification of environmental flow requirements of freshwater biota through further monitoring and experimentation should be considered a top research priority by conservation scientists.

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Table 2.1 Estimated coefficients of Poisson regression model for juvenile steelhead recruitment (n = 500).

Coefficient[1]	Mean	Std. Dev.	95% CI	
Constant	**-1.993**	**0.291**	**-2.649**	**-1.449**
Habitat Suitability	**0.432**	**0.209**	**0.019**	**0.826**
Median Spring Flow	0.231	0.162	-0.092	0.548
Road Density	-0.024	0.731	-1.338	1.360
Vineyard Use (%)	**-0.376**	**0.135**	**-0.651**	**-0.102**

[1] Model coefficients that explain observed variation in recruitment (shown in bold) have estimates with 95% credible intervals that do not encompass 0.

Table 2.2 Estimated coefficients of logistic regression model for juvenile steelhead over-summer survival (n = 437).

Coefficient[1]	Mean	Std. Dev.	95% CI	
Constant	**-0.981**	**0.305**	**-1.592**	**-0.391**
Median Runoff (mm)	**1.597**	**0.055**	**1.490**	**1.705**
Low Flow Days (daily runoff < lower quartile)	**-0.004**	**0.001**	**-0.006**	**-0.002**
Summer Density (fish·m^{-1})	**-0.031**	**0.006**	**-0.042**	**-0.020**
Habitat (high suitability)	**0.267**	**0.084**	**0.104**	**0.432**
Road Density (km·km^{-2})	0.422	0.678	-0.975	1.788
Vineyard (% cover)	-0.226	0.131	-0.499	0.024

[1] Data from all paired initial-final summer counts, excluding observations where the initial summer count was 0. Model coefficients that explain variation in survival (shown in bold) have estimates with 95% credible intervals that do not encompass 0.

Figure 2.1 Map showing the locations of juvenile steelhead trout survey reaches (black boxes) in tributary streams to the Russian River, California, USA. The inset map indicates the study location within California and highlights the coastal region that share similar mediterranean-climate and land-use pressures.

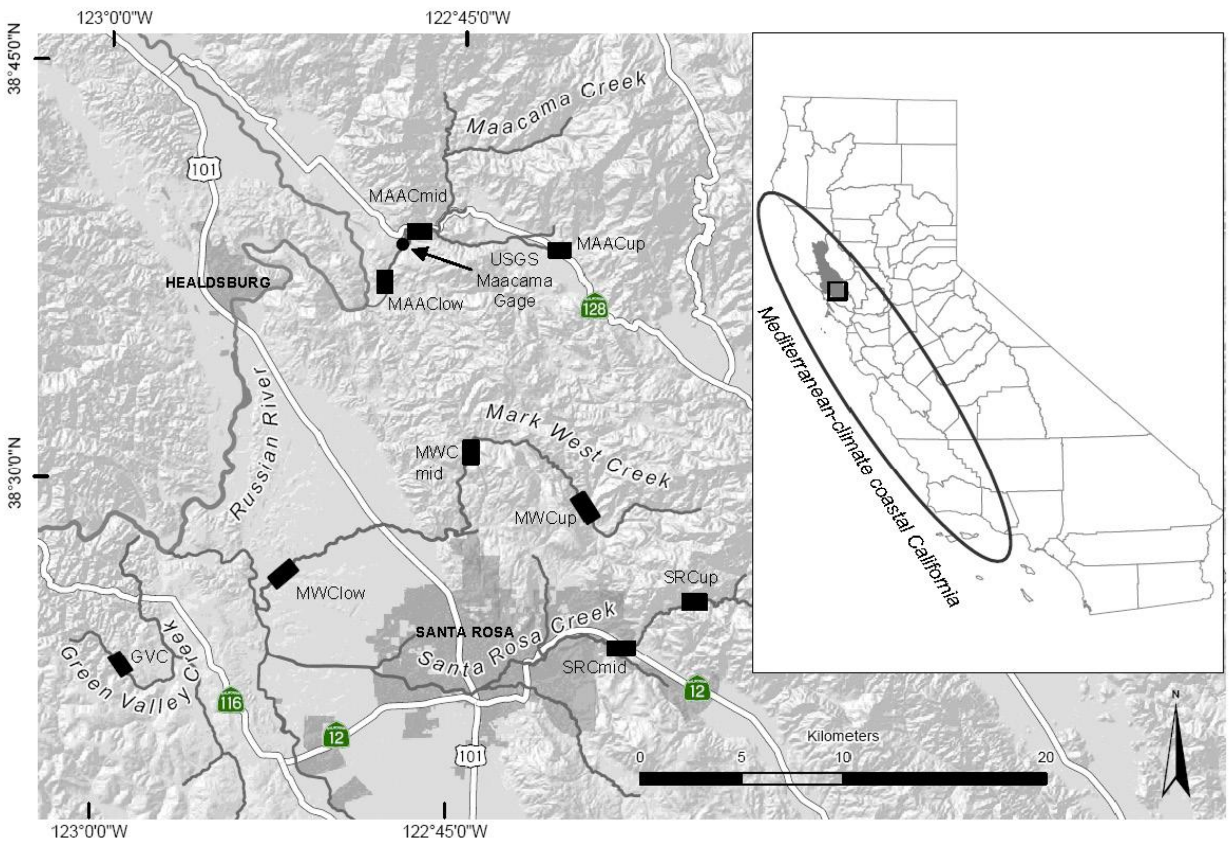


Figure 2.2 Typical spring-summer hydrograph of streams in coastal California with discharge plotted on log-transformed (lower box) and linear (upper box) axes, illustrating dry season flow patterns and timing of fish sampling in early and late summer. Gray bars denote life history stages for salmonids in the study region. The two stream flow variables used in the model are median spring stream flow (based on daily flow values between 1 March and 30 June, normalized by drainage area) and summer low-flow days (the number of days between 1 July and 30 September that fall below the lower quartile of daily summer flows observed at that reach over the period of record). The median spring flow and low-flow variables capture the relative flow magnitude and timing of flow recession, respectively, for each year of the study period.

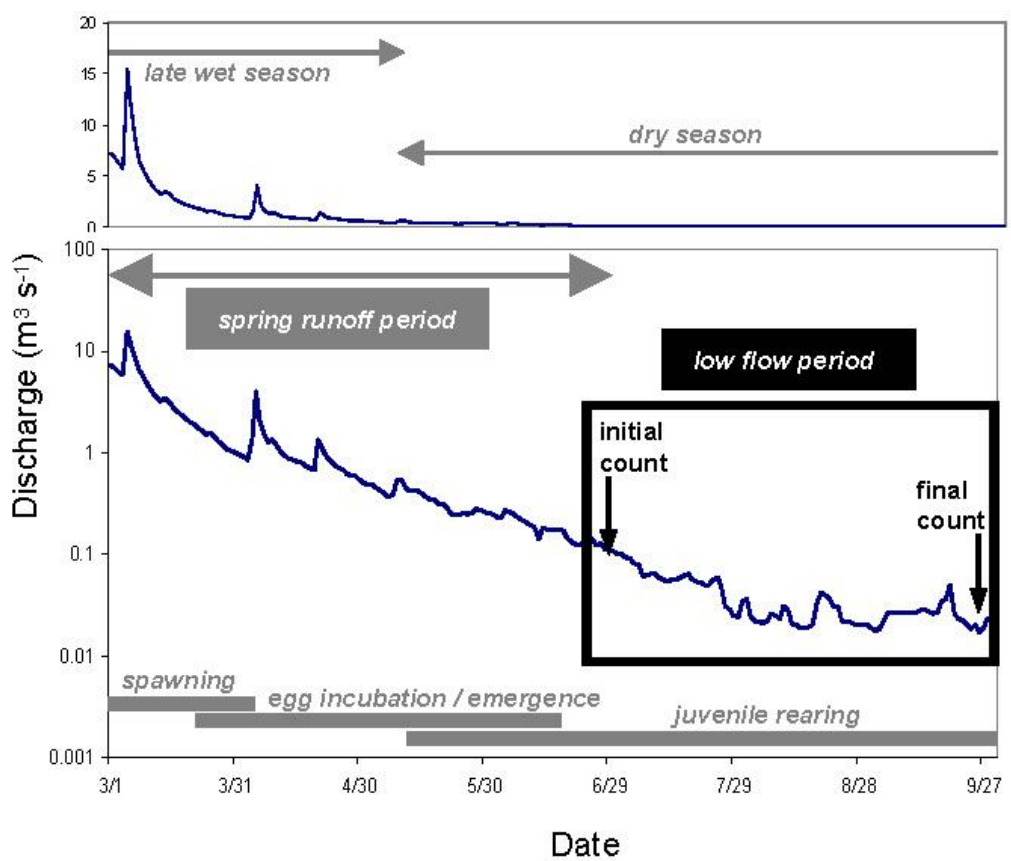


Figure 2.3 Observed early summer and late summer juvenile steelhead trout densities (A) across years and (B) across reach sites. Whiskers indicate one SD from the mean. Data are from repeated fish surveys conducted in isolated habitat units (n = 54–59 per year) in the early and late summer over a 9-year period (1994–2002).

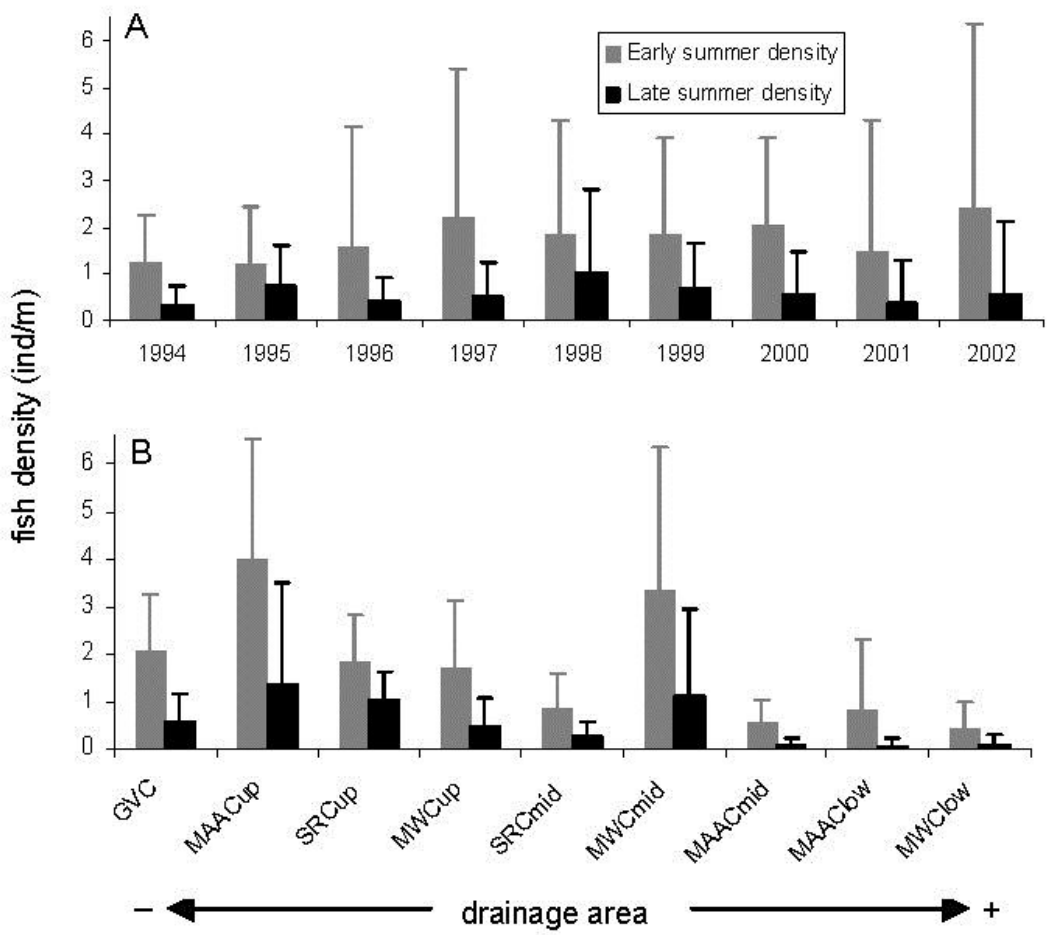


Figure 2.4 Multiplicative effects on recruitment and survival for median spring flow, the number of summer low-flow days, habitat suitability, road density and vineyard use. Mean multiplicative effects (with 95% CI) indicate the predicted magnitude of change in the response variable over the range of each explanatory variable, based on the regression models for recruitment (Table 2.1) and survival (Table 2.2). A multiplicative effect of one corresponds to no detectable change in recruitment or survival in response to the explanatory variable.

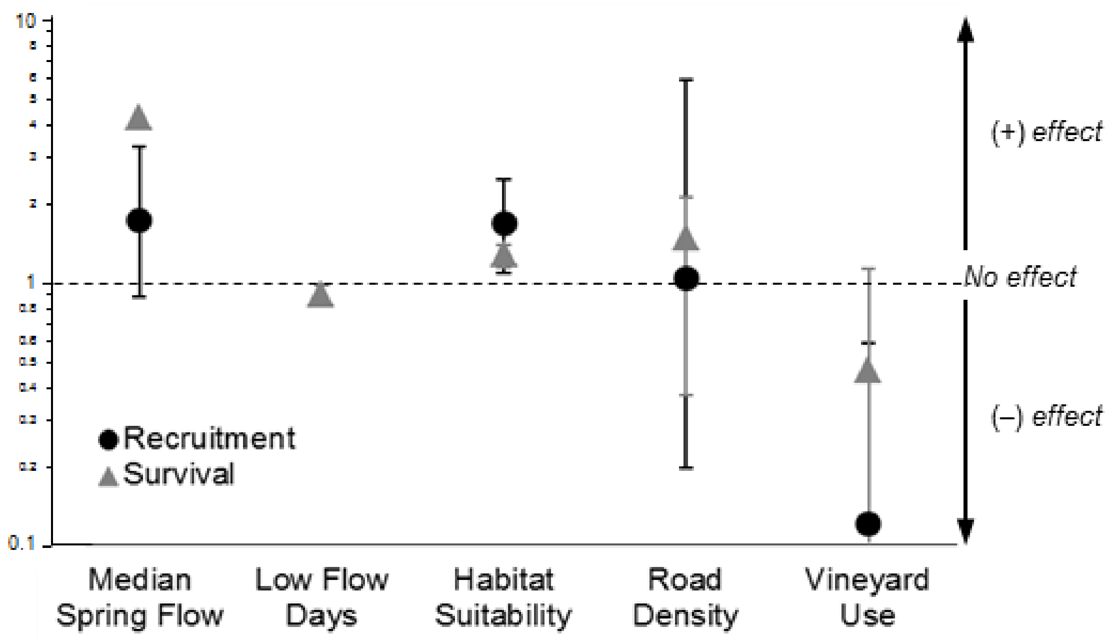
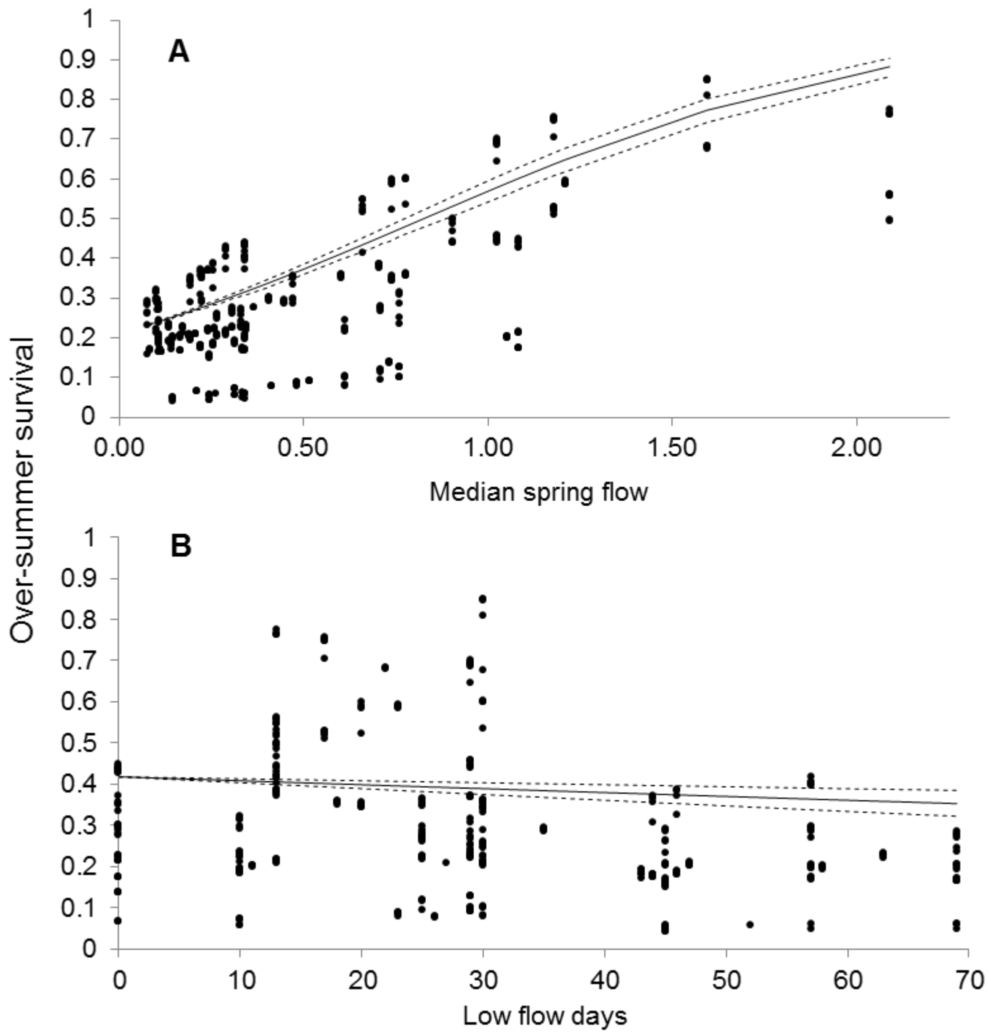


Figure 2.5 Predicted relationship of over-summer survival to (A) median spring runoff and (B) the number of summer low-flow days, conditional on the mean value for all other variables. The logistic regression relationship with 95% credible intervals plotted with unit-scale survival rates (n = 437) shows increasing survival with increasing annual spring runoff and decreasing survival with increasing number of low-flow days. Low-flow days quantifies the duration that mean daily summer season (July 1–Sept. 30) discharge fell below the lower quartile value, calculated for each reach over the period of record (1994–2002).



CHAPTER 3

AN INTEGRATED FRAMEWORK FOR STREAMFLOW MANAGEMENT IN MEDITERRANEAN-CLIMATE CALIFORNIA, U.S.A

Theodore E. Grantham, Adina M. Merenlender, and Vince H. Resh

An integrated framework for streamflow management in
Mediterranean-climate California, U.S.A

ABSTRACT

In Mediterranean and other water-stressed climates, water management is critical to the conservation of freshwater ecosystems. To secure and maintain water allocations for the environment, integrated water management approaches are needed that consider ecosystem flow requirements, patterns of human water demands and the temporal and spatial dynamics of water availability. Human settlements in Mediterranean climates have constructed water storage and conveyance projects at a scale and level of complexity far exceeding those in other, less seasonal climates. As a result, multiple ecological stressors associated with natural periods of flooding and drying are compounded by anthropogenic impacts resulting from water infrastructure development. Despite substantial investments in freshwater ecosystem conservation, particularly in California, U.S.A., success has been limited because the scales at which river management and restoration are implemented are often discordant with the temporal and spatial scales at which ecosystem processes operate. Often, there is also strong social and political resistance to restricting water allocation to existing consumptive uses for environmental protection purposes. Furthermore, institutions rarely have the capacity to develop and implement integrated management programs needed for freshwater ecosystem conservation. We propose an integrated framework for streamflow management that explicitly considers the temporal and spatial dynamics of water supply and needs of both human and natural systems. This approach makes it possible to assess the effects of alternative management strategies to human water security and ecosystem conditions and facilitates integrated decision-making by water management institutions. We illustrate the framework by applying a GIS-based hydrologic model in a Mediterranean-climate watershed in Sonoma County, California, U.S.A. The model is designed to assess the hydrologic impacts of multiple water users distributed throughout a stream network. We analyze the effects of vineyard water management on environmental flows to (i) evaluate streamflow impacts from small storage ponds designed to meet human water demands and reduce summer diversions, (ii) prioritize the placement of storage ponds to meet human water needs while optimizing environmental flow benefits and (iii) examine the environmental and social consequences of flow management policies designed to regulate the timing of diversions to protect ecosystem functions. Spatially explicit models that represent anthropogenic stressors (e.g. water diversions) and environmental flow needs are required to address persistent and growing threats to freshwater biodiversity. A coupled human–natural system approach to water management is particularly useful in Mediterranean climates, characterized by severe competition for water resources and high spatial and temporal variability in flow regimes. However, lessons learned from our analyses are applicable to other highly seasonal systems and those that are expected to have increased precipitation variability resulting from climate change.

INTRODUCTION

Mediterranean-climate regions are concentrated centers of both human populations and agricultural production. Consequently, competition for water in these areas is among the highest in the world (Gasith and Resh 1999). Human needs for fresh water in Mediterranean-climate areas are further complicated by unpredictable, annual water supplies and the seasonal disconnect between when water is available and when demand is highest. Unpredictable annual precipitation and limited water availability during the dry season have resulted in extensive water infrastructure development, which complicate efforts to restore and manage freshwater ecosystems. Thus, the global challenge of balancing ecosystem integrity with societal water needs (Baron et al. 2002) is acute in Mediterranean climates.

The conservation of freshwater ecosystems requires new water management approaches that consider both societal and ecosystem needs in an integrated fashion (Wallace et al. 2003). Integrated water resources management is particularly needed in Mediterranean- climate regions, where the conservation of stream ecosystems requires the modification or curtailment of human water use practices. In these regions, and in other areas of the world with water-stressed systems, the maintenance of natural flows through environmental water allocations is essential to the conservation of freshwater ecosystems (Arthington et al. 2006; Dudgeon et al. 2006; King and Brown 2006). Yet, substantial scientific, social and institutional challenges continue to hinder the implementation of ecologically sustainable water management.

In this article, we highlight the importance of integrated streamflow management in Mediterranean climates and describe a framework that takes into account the complex dynamics of water availability, human water demands and ecosystem needs at appropriate spatial and temporal scales. First, we describe patterns of water resources development in Mediterranean-climate regions and their associated ecosystem effects. Second, we discuss the challenges of river ecosystem management and protecting environmental flows in these regions. Third, through an example from a Mediterranean-climate California watershed, we demonstrate how a coupled human–natural system approach to river management makes it possible to meet agricultural water demands while minimizing impacts to environmental flows, as well as to evaluate the potential consequences of alternative water management policies. Finally, I highlight its applications for managing water resources for ecosystem and human needs in other seasonal climates.

MEDITERRANEAN-CLIMATE REGIONS

Mediterranean-type climates occur on all continents, extending between 30 and 40° latitude both north and south of the equator. Most of this climate type is located around the Mediterranean Sea. On the Pacific coast of North America, the Mediterranean-climate region extends from southern Oregon to northern Baja California. Other parts of the world with a Mediterranean climate include parts of west and south Australia, the south-western Cape region of South Africa, and the central Chilean coast (Ashmann 1973). Mediterranean climates, which are often westerly positioned, are the result of a symmetrical atmospheric circulation that produces a characteristic pattern of cool, wet winters and dry, hot summers (Ashmann 1973). Moderating oceanic influences keep winter temperatures mild, with mean monthly temperature minima between 8 and 12℃; summer maxima typically range from 18 to 30℃ (Gasith and Resh 1999).

Mediterranean-climate regions exhibit predictable, seasonal patterns of rainfall and drought. Most of the annual precipitation is concentrated over a few months in the winter, and there is often little to no rain from late spring to early fall. Compared to other regions with similar total annual rainfall, the amount and timing of precipitation within the wet season is also highly variable between years, leading to an extremely uncertain renewable supply of fresh water (Merenlender et al. 2008). In addition to high temporal variability in precipitation, Mediterranean systems often have complex tectonic and geologic conditions that result in high levels of spatial heterogeneity in streamflow regimes within river basins (Conacher and Sala 1998).

Mediterranean regions are highly suitable for human habitation and have settlements that can be traced back to the earliest civilizations. It is not surprising that humans found these areas attractive given the rich soils, abundant sun and long growing season. Today, Mediterranean-climate regions continue to support concentrated human populations and are global centers of agricultural production. In Mediterranean-climate regions, freshwater lakes are rare and ground water is often restricted to river flood plains, so human settlements rely extensively on streams and rivers to meet their water demands for agriculture, industry and domestic consumption (Gasith and Resh 1999). The withdrawal of water for agricultural irrigation in these regions typically represents the vast majority of total human water use (e.g. up to 80%), although water withdrawals for urban use has been increasing in Mediterranean-climate regions with growing populations such as California (Konieczki and Heilman 2004) and in countries such as Spain, Morocco and Israel [World Commission on Dams (WCD) 2000].

Most of the human water demands in Mediterranean-climate areas occur in the dry summer months, when water is required for agricultural irrigation yet precipitation is rare. Thus, the asynchronous timing of water availability and demands, together with high interannual variability of Mediterranean river flows, have led to the development of large-scale water storage and irrigation projects to maintain reliable water supply. In fact, the extensive manipulation of rivers to provide reliable access to water is a defining characteristic of Mediterranean systems (Kondolf and Batalla 2005). In California, for example, every major stream has been affected by the construction of dams and reservoirs to increase water supply security for agricultural and urban water users (Moyle 2002).

While water infrastructure development has increased water security for human settlements, it has also substantially altered the natural flow dynamics of river and streams. The hydrologic alterations to rivers associated with large water projects are well documented in Mediterranean climates and throughout the world. Large dams are specifically intended to alter the natural distribution and timing of streamflow (Poff et al. 1997), and disruption to natural flow regimes occurs both upstream and downstream of dams and major diversions (e.g. Graf, 1999; Cowell and Stoudt 2002; Nilsson et al. 2005; Richter and Thomas 2007). Water infrastructure development also often entails the transfer of water across natural geographical boundaries. These inter-basin transfers augment water supplies in some basins while dewatering the basin of origin, altering natural flow patterns across broad geographical regions (Davies et al. 1992).

In areas not served by large reservoirs, small water projects are common, including surface water diversions and ground water pumping. Direct withdrawal of water from streams can result in decreased flows by more than 90% locally and can produce significant cumulative downstream effects (Deitch et al. 2009a). As alternative to on-demand withdrawals from streams or ground water, small water storage basins (also referred to as farm ponds) are often built that are filled from surface water diversions and run-off captured in the winter for later use in the growing season. The hydrologic impacts of small storage ponds are less well studied than large reservoirs, although they are likely to have similar effects on downstream flows albeit on a smaller scale.

BIOLOGICAL RESPONSES TO MEDITERRANEAN CLIMATES

In Mediterranean climates, seasonal fluctuations in streamflow, as well as episodic disturbance events (e.g. interannual floods and drought), have a dominant influence on freshwater ecosystem structure and function (Resh et al. 1988). Mediterranean stream biota experience the sequential occurrence of extreme abiotic disturbance (e.g. winter floods), followed by a period of increased biotic interactions (e.g. predation and competition for food) as lotic habitats become more lentic, and finally more abiotic pressures as drying and loss of hydrologic connectivity occurs (Gasith and Resh 1999; Bonada et al. 2006).

The influence of climate variability on ecosystem structure and functions has been studied across all trophic levels in diverse Mediterranean-stream systems. Most studies of climatic influences on biota have focused on macroinvertebrates, demonstrating strong effects of seasonal and interannual precipitation patterns on the composition and abundance of benthic species (e.g. Bêche et al. 2006; Bonada et al., 2006; Elron et al. 2006; Bêche and Resh 2007; Bonada et al. 2007b; Daufresne et al. 2007; Dewson et al. 2007; Rader et al. 2008). Hydrologic variability has also been demonstrated to have strong effects on fish assemblages (e.g. Bernardo et al. 2003; Magalhães et al. 2007; Bêche et al. 2009), primary productivity (e.g. Marks et al. 2000; Schemel et al., 2004) and food web structure (e.g. Power et al. 2008; Strayer et al. 2008).

In Mediterranean climates, native biota have developed a variety of mechanisms to tolerate the environmental stressors of seasonal flooding and drying and associated changes in habitat conditions (Fox and Fox 1986). For example, Bonada et al. (2007a) found that, in comparison with temperate-climate streams, macroinvertebrates from Mediterranean streams tended to have life history traits that provided greater resistance to droughts and improved ability to recovery from disturbance. Mediterranean stream fish species also have life history traits that enhance their ability to cope with hydrologic variability, including short lifespans, rapid growth rates and high fecundity (Ferreira et al. 2007), as well as behavioral adaptations to respond to flow-related stressors, such as migratory movement to refugia during drought periods (Magoulick and Kobza 2003).

There is also evidence that natural hydrologic disturbance plays an important role in structuring species assemblages in Mediterranean streams. For example, Marchetti and Moyle (2001) demonstrated that native fishes in a regulated California stream responded positively during wet years when flow conditions were more similar to their natural regime. In drier years, when dam releases are reduced in the summer, lower flows and higher temperatures created habitat conditions that were less suitable for native fish species and more favorable for non-native fishes. Episodic, bed-souring winter flows have also been shown to be important for structuring river ecosystems, influencing algal biomass, invertebrate communities and trophic interactions that persist through the low-flow season (Power et al. 2008).

Despite the importance of natural hydrologic disturbance in mediating biotic interactions and community structure, Mediterranean freshwater ecosystems are highly susceptible to impacts from water management operations and other anthropogenic disturbances (Alvarez-Cobelas et al. 2005). Not only does water infrastructure development tend to be extensive in Mediterranean-climate regions (Kondolf and Batalla 2005), but the highly adapted nature of native aquatic species to natural flow variability (Lytle and Poff 2004) may make them particularly vulnerable to activities that affect natural flow patterns. Storage reservoirs and dam operations tend to reduce flow variability (Poff et al. 2007) and diversions are capable of reducing flows necessary to maintain stream habitat conditions. The vulnerability of freshwater species to flow regime impacts combined with the pervasive extent of water resources development in Mediterranean-climate regions thus makes water management a key issue for freshwater ecosystem conservation (Richter et al., 2003; Dudgeon et al., 2006).

Water management operations, such as diversions, dams and flow regulations, interfere with fundamental hydrologic processes that control habitat structure, the intensity and frequency of scouring floods, floodplain interactions and water quality conditions (Bunn and Arthington 2002). Water management activities commonly associated with ecosystem impacts can be grouped into four broad categories: (i) water diversions, (ii) impoundments, (iii) dam operations and (iv) inter-basin transfers. The ecological effects of these activities on freshwater ecosystems are well documented (reviewed by Poff et al. 1997; Bunn and Arthington 2002; Murchie et al. 2008; Haxton and Findlay 2008; summarized in Table 3.1).

Although less well studied than dams and diversions on large rivers, water management of small, unregulated streams can also impair ecologically relevant flow regime characteristics. In stream catchments where water demand is high, the local and cumulative impacts of surface water diversions have the potential to accelerate drying over extended stream reaches and to reduce habitat availability for aquatic species (McKay and King 2006; Spina et al. 2006; Deitch et al. 2009a). The ecological responses to decreased low flows remain poorly understood, but artificially reduced flows from water extractions are likely to result in shifts in the abundance, diversity and composition of both invertebrate and fish species (Dewson et al. 2007).

In Mediterranean and other climatically variable systems, flow regime alterations from water management operations have played a dominant role in the decline in freshwater biodiversity (Dudgeon et al. 2006). In California, water diversions have been identified as the most significant human activity negatively affecting fish diversity, where 40% of native fish populations have been driven to extinction or are in serious decline (Moyle 2002). Similar patterns of freshwater ecosystem degradation are observed in Mediterranean Europe, where water development has contributed to a 50–100% decline of native fish species abundance since the beginning of the 20th century (Aparicio et al. 2000).

FRESHWATER CONSERVATION AND STREAMFLOW MANAGEMENT

Freshwater ecosystems have experienced widespread degradation at a global scale and generally remain poorly protected despite persistent and growing threats (Dudgeon et al. 2006). In Mediterranean-climate regions such as California, conservation and restoration programs have failed to reverse the trend of freshwater biodiversity loss or achieve substantive protections of environmental flows for endangered aquatic species, despite investments of more than 2 billion dollars ($US) to date on restoration projects alone (Kondolf et al. 2007).

In our view, the limited success of freshwater ecosystem conservation in California, as well as in other Mediterranean-climate regions, is largely attributed to three important factors. First, the temporal and spatial scales at which conservation and restoration programs are conducted are often discordant with the scales at which ecosystem processes operate. Second, there is strong social and political resistance to restricting existing consumptive uses for environmental flow protection purposes. Finally, institutions do not have the capacity to develop and implement integrated programs required for sustainable freshwater management.

Problems of scale

The effectiveness of freshwater conservation has been compromised by the limited extent at which conservation programs and restoration treatments are actually implemented. In Mediterranean and other climate types, restoration has traditionally been conducted at the reach scale and been focused on the recovery of form and pattern, and thus produced limited ecological benefit when fundamental ecosystem processes (e.g. watershed hydrologic functions) have been altered (Wohl et al. 2005; Kondolf et al. 2006). The fragmented approach to restoration is highlighted in a recent assessment of river restoration projects in California by Kondolf et al. (2007), who found that of the projects surveyed, <10% considered a broader watershed management plan during site selection and project design. They conclude that most restoration projects fell short of restoring dynamic watershed processes and thus probably are of limited ecological value.

In contrast to the predominant approach to restoration described earlier, many restoration scientists have now come to understand that the restoration of an acceptable range of variability of process is more likely to succeed than restoration aimed at a static state that neglects environmental variability (Richter 1997; Wohl et al. 2005). This latter approach requires increasing the spatial scales at which restoration programs are commonly planned and implemented.

Finally, the conservation of Mediterranean freshwater ecosystems not only requires consideration of spatial scale, but also must address the challenges associated with the temporal variability of precipitation and streamflow patterns. Because Mediterranean systems are characterized by long-term disturbance regimes, episodic and extreme flood events in Mediterranean rivers can significantly alter river morphology and vegetation patterns. Depending on the elapsed time since the last major flood, Mediterranean rivers may exhibit strikingly different characteristics in morphology, vegetation patterns and biological community composition (Hughes et al. 2005). Because our knowledge of the range of natural river states is often limited, it is difficult to establish meaningful baseline conditions to set restoration objectives (Wohl et al. 2005). Furthermore, natural climatic variation causes fluctuations in the abundance and distribution of species of concern, which obscures larger-scale trends and human-related factors responsible for population declines (Ferreira et al. 2007).

Challenge of balancing human and natural system water needs

The effectiveness of freshwater ecosystem conservation has also been limited in Mediterranean climates because of the high competition for water resources. Conservation of freshwater biodiversity often requires making trade-offs between environmental and human water uses (Baron et al. 2002; Poff et al. 2003; King and Brown 2006). To protect water allocations to stream ecosystems, minimum flow thresholds are often imposed to restrict human consumptive uses. Such measures are often critical for maintaining natural flows in streams, yet often stimulate significant social, political and economic friction. As a result, environmental water allocations to improve ecological conditions are rarely considered in river restoration practice, which remains mostly focused on habitat improvements such as planting riparian vegetation, reducing sediment and reconstructing stream channels (Christian-Smith and Merenlender 2010). Our experience from California suggests that when conservation efforts do address environmental flows, a strong connection with endangered species populations must be established. Although changes in dam operations are increasingly considered for the recovery of aquatic ecosystems (Richter and Thomas 2007), environmental flow protections remain weak or nonexistent in the vast majority of rivers and streams affected by diversions and dams. Furthermore, the integration of environmental flow allocations in water management has largely been focused on regulated rivers, while strategies for protecting environmental flows in smaller, unregulated streams that are affected by water diversions have received far less attention.

Human population growth and climate change are expected to impose increasing pressures on freshwater resources, placing even greater constraints on aquatic species conservation (Postel et al. 1996). In the Mediterranean basin, for example, most of the available water resources have already been developed, while population growth and urbanization are expected to significantly increase human water demands (Araus 2004). In addition, global climate change is likely to lead to increased temperatures and changes in the amount and timing of rainfall (Mannion 1995), further reducing regional human water security. In Mediterranean systems, as well in other water-stressed regions, balancing human and ecological water needs remains a daunting challenge for freshwater ecosystem conservation.

Institutional constraints

Governmental institutions responsible for freshwater ecosystem protections and allocations of water resources are often poorly equipped to implement the types of integrated approaches required for sustainable water management. For example, increasingly larger scales of water infrastructure development in Mediterranean regions has been coupled with increasing scales of governance over these systems, which often lead to distinct regional, national and international administrative systems of management control. The increasing scales of water management institutions have enabled the construction and operation of large-scale water projects that have allowed for economic growth in areas that otherwise would be constrained by water scarcity. However, the development of large-scale institutions has also led to the fragmentation of water management authority. In California, for example, distinct authorities are responsible for regulating water quality (to protect beneficial human uses), managing historic water rights (to protect economic interests of individual water users) and, more recently, ensuring aquatic species protections. There is also a division of control among institutions depending on the origin of fresh water being used. For example, in California, the State Water Resources Control Board (SWRCB) regulates surface water diversions but has no authority over ground water use, despite the fact that, in many cases, regional surface water and ground water sources are hydrologically connected (Sax 2003).

When responsibility for social, economic and environmental protections are partitioned in this way, an action of one institution consistent with its legal mandate can have unanticipated effects on the others. For example, legal stream withdrawals by private landowners may cumulatively impact the amount and timing of water delivery to city or regional water authority and/or environmental flows necessary to support endangered aquatic species. In other contexts, flow releases from dams to meet environmental flow regime targets can reduce water security for farmers or affect trans-boundary water agreements. Inevitably, the multiple and often overlapping scales of jurisdiction, coupled with the fragmentation of governance structures, impede institutions from performing the fundamental tasks of integrated water management. Consequently, broad-scale assessments of water availability and uses, coordinated monitoring and decision-making, planning and implementation are often suboptimal (Davis 2007).

INTEGRATED FRAMEWORK FOR STREAMFLOW MANAGEMENT: EXAMPLES FROM THE RUSSIAN RIVER BASIN, CALIFORNIA, U.S.A.

In the light of the problems facing freshwater conservation in Mediterranean-climate regions, we propose an integrated framework for streamflow management that explicitly considers the temporal and spatial dynamics of water supply and the needs of both human and natural systems and that is intended to facilitate analysis and decision-making at broad geographical (e.g. basin) scales. This framework relies on a GIS-based hydrologic model to: (i) quantify patterns of water availability at scales relevant to ecosystem needs, (ii) represent the timing, magnitude and location of human water demands in relation to ecosystem flow requirements and (iii) calculate the local and cumulative impacts of alternative water management strategies. Our modeling framework addresses river ecosystems that are characterized by high variability in flow conditions and subject to population and landuse pressures that require year-round water supplies. This model is particularly well-suited to assess decentralized water management systems, such as free-flowing rivers and streams that are affected by a spatially distributed network of water users. Through an example from a northern California watershed, we demonstrate that despite the complexity and pressures on streams in Mediterranean climates, it is possible to reduce potential ecosystem impacts while addressing human water needs.

Study area

The Russian River basin is a large coastal watershed (3900 km^2) in northern California, where 11 incorporated cities ranging from the densely populated Santa Rosa in the south (population 150 000 in 2000) to more rural communities in the north. The basin is also one of California's premium wine-grape growing regions and supports a thriving tourist economy. As in other Mediterranean climates, many smaller and upland watersheds are increasingly being used to grow high-value agricultural crops, such as vineyards, olive trees and avocados (Merenlender 2000). Two large reservoirs in the basin supply most of the urban water demand in the region, while vineyards and other agriculture rely almost entirely on locally available surface and ground water resources.

The Russian River basin is home to three salmonid species listed under the federal Endangered Species Act: the central California coast Coho salmon (*Oncorhynchus kisutch*), central California coast steelhead trout (*Oncoryhnchus mykiss*) and California coast Chinook salmon (*Oncorhynchus tshawytscha*). Although several factors have contributed to population declines, flow regime and water quality alterations resulting from water management are considered a primary threat to California's salmonid species (Moyle 2002).

California salmonids are highly adapted to the natural flow regime of coastal rivers and streams. Lower-velocity, winter baseflows between storm events allow adult salmonids to migrate from the ocean to spawning grounds and provide suitable hydrologic conditions for egg incubation. Winter peak flows are important for maintaining appropriate sediment distributions for spawning and preventing vegetation encroachment into the stream channel. In the spring, streamflows maintain hydrologic connectivity, allowing for juvenile out-migration and providing food resources via downstream drift. Summer flows maintain connectivity until streams approach or become intermittent, whereby pools continue to provide over-summering habitat until flows resume again in the fall (Kocher et al. 2008). These lower spring and summer flows are critical for maintaining suitable habitat for juvenile rearing, and may be vulnerable to impacts from diversions because water demands are greatest during the dry season.

An increase in catchment storage capacity through small distributed storage ponds (with average volumetric capacities of 50 000 m^3) provides one alternative to pumping water on demand from rivers or ground water during the dry season. Where local water demands are met by direct surface water diversions, the ability to irrigate from stored winter water has the potential to ameliorate the impacts on summer environmental flows. However, consideration must also be given to potential impacts on winter flows, because storage ponds are expected to reduce downstream flows until they are filled. Yet little is known about the extent to which distributed networks of small water storage projects lead to individual and cumulative impacts on environmental flows (Merenlender et al. 2008). Most studies on trade-offs between water storage benefits and environmental flows have focused on large dam operations (e.g. Richter and Thomas 2007) and not on decentralized water management.

The volumes of water required for vineyard irrigation typically represent a small fraction of the total water availability in the winter months (Deitch et al. 2009b). Therefore, the specific challenge in this system is to determine the number, size and locations of winter storage ponds needed to offset summer water demands without significantly impacting winter environmental flows. Our model is designed to examine the effects of alternative water storage scenarios and makes it possible to prioritize sites where storage will provide the greatest benefit to summer environmental flows while considering potential winter flow impacts.

Another important application of our model is to assess the effects of alternative water policies on the timing and magnitude of allocations to meet human and ecosystem needs. Where environmental flow protections are needed to conserve freshwater ecosystems, the model provides a tool to understand the consequences of flow protections on both ecological and human systems across spatial and temporal scales.

Methods

We use a GIS-based (ArcGIS, version 9.2; ESRI 2006) hydrologic model developed by Merenlender et al. (2008) to examine the effects of small water storage ponds on streamflow regimes throughout the year, in a 16 km^2 catchment in Sonoma County, California. The model estimates stream discharge (m^3s^{-1}) at all points in a drainage network based on records from a nearby USGS gage station (Maacama Creek #11463900, in eastern Sonoma County, CA, U.S.A.), scaled according to watershed area and annual precipitation differences. For this exercise, we used flow data from a normal rainfall year (e.g. 1966, a year with median annual discharge), although any hydrograph may be specified in the model.

We digitally mapped hypothetical storage ponds on the landscape, specifying their coordinate locations and volumes. The model was then run to route flows through the catchment with specifically placed storage ponds, which are assumed to capture all upstream run-off until they are filled. For each scenario, the model calculates the flow impact (e.g. percentage of flow removed from stream compared to unimpaired flow) below each reservoir. These effects are then propagated down the stream network.

The downstream impact of a pond depends on its volume and location in the drainage network, and pond effects can be described by the degree of flow impairment (e.g. percentage of natural flow removed from channel downstream), impact length (e.g. length of channel downstream that flows remain impaired) and duration of impairment (e.g. number of days in which flows are impaired as the pond fills). To compare the impacts of different water storage sites, we developed an impact index that aggregates each of these pond impact types, based on the following metric:

$$\begin{aligned}&Impact\ Index = \\ &No. of\ impact\ days \times \sum_1^n (Segment\ length\ \times Percentage\ flow\ impairment)\end{aligned} \quad [1]$$

for n flow-impaired, 10-meter segments below the storage pond location. This impact index is useful because it captures both the spatial and temporal extent of downstream effects on winter flows.

To consider the environmental and human-system effects of surface water storage within the study catchment, we applied the model in three ways. First, we placed three hypothetical storage ponds of equal volume in the catchment and calculated their individual and cumulative downstream impact indices. This application of the model demonstrates how the impacts of storage ponds are influenced by upstream area and downstream drainage network configuration. The example also serves to illustrate how multiple ponds interact to produce cumulative effects on streamflow.

In the second example, we placed hypothetical ponds on landholdings with existing vineyards, setting the storage volume of each pond equal to the estimated annual water demand of that vineyard (e.g. 0.2 m^3m^{-2}). We then calculated the impact index of each reservoir and compared them to the benefits they provide in offsetting summer water demands in the catchment. Based on this benefit-to-impact ratio, we ranked each of the hypothetical ponds, illustrating how the strategic prioritization of storage projects could be achieved.

In the final example, we examined the potential consequences of a proposed California streamflow management policy on both winter and summer environmental flows [State Water Resources Control Board (SWRCB), 2007]. The policy is designed to protect winter environmental flows and restricts water diversions to periods when flow exceeds a threshold level necessary for upstream salmon migration. The flow threshold is defined as the minimum flow necessary for maintaining sufficient water depths for adult salmon passage. We ran the model for each of the ponds from the previous example, under the policy scenario where the timing and volume of diversions are restricted to protect winter environmental flows. We then assessed how implementation of the policy would modify winter flow impacts, storage volumes of winter water and potential demands on summer flows.

Example 1: effects of storage pond location on streamflow
As discussed previously, one solution to offset summer flow impacts is to capture water from winter flows into storage ponds for use during the irrigation season. However, the placement of ponds within a catchment requires consideration of impacts on winter environmental flows, which are important for fish migration and channel maintenance. The impacts of three hypothetical ponds placed in the study watershed are illustrated in Figure 3.1. All three of the ponds have the same storage capacity but have substantially different impacts on downstream flows because of differences in site location and upstream drainage area.

The model results show that Pond A has a greater downstream impact on winter flows than Pond B (Figure 3.1). Because Ponds A and B have the same catchment area (3 km2), the number of days to fill is the same (64 days), after which spillover occurs and flows are unimpaired. However, flows from the tributary downstream of Pond B reduce the impact of the reservoir compared to Pond A. Therefore, the impact index of Pond A (152 km day) is nearly three times as great as that of Pond B (59 km day).

The impact index of Pond C (171 km day) is greater than those of both A and B. Pond C has the same volume as Ponds A and B, but fills more rapidly (46 days) because of the greater upstream drainage area. However, Pond C has a much larger downstream effect than Ponds A and B because it captures run-off from a larger drainage area, cutting off significant flow contributions to the channel downstream.

The model indicates that early-winter streamflow at the catchment outlet is impaired by as much as 90% when the cumulative effects of all three ponds are analyzed. The impact index of all three ponds together is 446 km day, which is substantially larger than the sum of the impacts of the three ponds considered separately (approximately 382 km day), illustrating the potential for multiple ponds to interact and produce significant cumulative effects on streamflow.

Example 2: targeting sites for storage by optimizing environmental flow benefits
Strategic placement of storage ponds across the landscape could theoretically reduce or eliminate summer water withdrawals, protecting environmental flows in the dry season. However, as

illustrated in Example 1, ponds have the potential to impact winter environmental flows. Therefore, the allocation of storage throughout a basin should consider the trade-offs associated with the benefits and impacts of specific pond locations. When we run the model to evaluate the effects of 14 hypothetical storage ponds, sized to meet the water demands of the surrounding vineyard parcel, we again find that individual impacts vary depending on their storage capacity, upstream drainage area and location in the stream network (Figure 3.2; Table 3.2). The 14 ponds range in volume from 10 000 to 150 000 m^3 and have upstream drainage areas of 0.04–1.2 km^2. The impact indices range substantially, from approximately 6 km day (Pond B) up to 105 km day (Pond H). When the individual ponds are ranked based on their benefit-to-cost (% of catchment storage-to-impact index) ratio (Table 3.2) and plotted against the proportion of water demand for the catchment, we see that approximately 70% of basin water demand could be offset by locating ponds at 6 of the 14 sites (Figure 3.3). The marginal increase in storage gained by adding more ponds decreases after the sixth highest-ranking site.

Example 3: consequences of environmental flow policies

As the final example, we examine the consequences of environmental flow policies designed to regulate the timing of diversions and reservoir filling. As described previously, the proposed environmental flow policy for northern California restricts diversions in the winter, such that impacts to downstream flows are greatly reduced. This is because, in most cases, flows exceed the minimum threshold only a few days over the winter. Because the diversion period is greatly reduced, the impact index calculated for all but one of the reservoirs under the proposed policy scenario drops to <0.05 km day (in contrast to the impact values shown in Table 3.2). While the policy significantly reduces potential impacts of storage to winter environmental flows, the restriction on diversions also reduces the total amount of water stored over the winter (Figure 3.4). A few of the ponds (e.g. G, H and L) fill completely, whereas others (e.g. D) receive no water because flows at the location failed to exceed the minimum threshold over the water year. Overall, approximately 60% of the total basin water demand would be met under the environmental flow protection policy if all 14 water storage ponds were installed, suggesting that agricultural irrigation needs would have to be met by diverting water during the summer months.

DISCUSSION

Applications and next steps

Despite the complexity and multiplicity of natural and anthropogenic stressors on river ecosystems in Mediterranean climates, our case studies suggest that it is possible to reduce potential ecological impacts and improve our management of water resources to meet both human and ecosystem needs. The model we propose supports an integrated approach to water management by accounting for the spatial and temporal variability in water availability, human water needs and environmental flow requirements. In addition, the model allows for the analysis of cumulative impacts, which are often difficult to quantify but may be a significant cause of ecosystem degradation in decentralized water management systems. Furthermore, the modeling framework can help to prioritize freshwater conservation efforts by evaluating the impacts and benefits of changes in water management practices on environmental flows. Finally, this framework makes it possible to assess the consequences of alternative policy scenarios and supports integrated decision-making by institutions responsible for water and freshwater ecosystem management.

Our model is focused on the management of surface flow in rivers and streams, because in Mediterranean climate regions they are the critical limiting resources for meeting human water needs and sustaining ecological functions. However, in some locations ground water is also important for meeting water needs, and the extraction of ground water has the potential to reduce surface flows and impact stream biota (Spina et al. 2006; Dewson et al. 2007). We expect future iterations of the model to incorporate surface water–ground water interactions to improve our predictions of streamflow and water availability. We also plan to incorporate additional complexity in the model by considering other drivers of human water use practices, including water rights, land values and local site topography.

Because our existing model does not include spatial variation in channel morphology and habitat conditions, an important future extension of the model will be to explicitly link the predictions of flow alterations with ecological impacts. Such an advancement would require a higher-resolution digital elevation model that captures changes in channel morphology within the drainage network, riparian vegetation structure, the spatial distribution of target species or assemblages and their responses to changes in flow (e.g. based on hydraulic preferences). However, these data are currently not available over the large spatial scales that the model is designed to analyze. Nevertheless, model impact predictions could inform reach-scale studies on the potential ecological effects of flow alterations through the application of instream habitat models, such as Physical Habitat Simulation (PHABSIM) or other environmental flow methodology (Tharme 2003). Ideally, such research efforts could be integrated with a broader framework to improve the knowledge of links between flow dynamics and biotic assemblage responses and guide water management decisions (e.g. Souchon et al. 2008).

Ultimately, the effectiveness of this model as a decision-support tool will be largely determined by the institutional capacity to conduct impacts analysis and develop management strategies at appropriate scales. This requires a formalized, integrated decision-making process and legitimate legal ⁄ political authority that are deficient both in Mediterranean and non- Mediterranean countries. Coordinated cross-governmental agency efforts will be needed to conduct catchment-scale assessments and more importantly to implement resulting planning priorities. Moreover, landowner participation and support will be critical for the success of this coupled human and natural systems approach to water management. Therefore, we agree that a collaborative approach encouraging participatory research is necessary, as has been described for developing environmental flow recommendations by Richter et al. (2006).

Conclusions

Freshwater ecosystem management and restoration, and environmental problem solving in general, will not result in the desired effects if the biological, physical or social impacts, and benefits are considered in isolation. Integrated approaches from multiple perspectives and disciplines are required (King et al. 2003). The approach we illustrate by the examples presented, and in our larger effort to increase our understanding of the coupled natural and human Mediterranean-climate watershed system, takes advantage of hydrologic modeling tools that make it possible to represent the spatial and temporal dynamics of human water use and ecological flow requirements. This approach would not be possible without the collaboration of hydrologists, economists, biologists and social scientists that has been fostered at our research institution (University of California, Berkeley) and that is increasingly being recognized as important in emerging interdisciplinary environmental science departments worldwide.

Likewise, interactions with landowners and policy makers through an active, participatory research program in northern California has been critical to our progress and is allowing us to move our models and exploration of hypothetical case studies from theory to practice. The early adopters of our decision-support tools are the rural landowners who have not been able to achieve water security or certainty in dealing with endangered species regulations. In contrast, resource institutions are more entrenched in their existing paradigms regarding impacts associated with multiple stressors and tend to avoid integrated approaches to environmental problem solving by relying on narrow definitions of their jurisdiction or regulatory responsibilities. At least in part, this is the result of a lack of resources to address the full complement of issues and cumulative impacts in particular.

As in other Mediterranean-climate regions, agriculture in California is responsible for around 80% of total water use. This has led many to argue that improvements in agricultural water use efficiency are necessary to meet the growing demands of other water users (e.g. urban and environmental) (Cooley et al. 2008). However, in our setting, the irrigation efficiency of vineyards is relatively high; therefore, improvements in efficiency are unlikely to yield significant gains in supply for other uses. Therefore, we must consider other ways to secure supplies for ecosystem needs. We acknowledge that the expansion of winter storage capacity to meet human water demands is a potentially controversial view given the ecological impacts caused by impoundments. In contrast to the position of reducing total consumptive uses through aggressive water-saving measures (e.g. fallowing agricultural lands and preventing further land development), we advocate a more pragmatic approach for managing the use of water. While recognizing the importance of water conservation efforts, we believe that there is probably some optimal storage capacity in a given watershed that will satisfy a significant proportion of human demands while maintaining adequate streamflows to protect environmental benefits. Some level of water storage development in Mediterranean-climate regions is not only appropriate, it is probably necessary for the long-term protection of freshwater ecosystems.

Our framework for streamflow management is relevant to freshwater ecosystem conservation in other climate regions. Global climate change is likely to result in greater uncertainty in natural water supplies in both Mediterranean and temperate climates (Araus 2004; Bonada et al. 2007a). Shifts in patterns of water availability may exacerbate current water management challenges arising from population growth and environmental degradation. In many regions, climate change will probably reduce the resilience of ecosystems to natural and human disturbances and further constrain freshwater ecosystem management. Thus, approaches to sustainable water management in highly variable-climate systems (such as Mediterranean regions) may become increasingly useful in other regions as the effects of climate change become evident.

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Table 3.1 Summary of hydrologic and ecological impacts of water management operations.

Water Management Activity	Hydrologic Impact	Ecological Impact
Water Diversions	Reduction in base flow magnitude and increase in duration	Reduced dilution capacity and water quality conditions favor pollution-tolerate species (2), concentration of aquatic organisms and increased biotic interactions (5), reduction or elimination of plant cover (5), loss of riparian species diversity (5)
	Acceleration of stream flow recession/drying	Failure of riparian seedling establishment (5), change in macroinvertebrate composition (2), decreased macroinvertebrate abundance (3)
	Lowered water table	Loss of riparian vegetation (2,5)
	Siphoning of surface waters	Aquatic species mortality from entrainment (1)
Impoundments	Reduction in frequency, duration and area of flooding	Reduction in inundation period of floodplain habitats used for spawning and foraging (1,5), decline in waterfowl species richness and abundance (1), shifts in riparian community composition (1), ineffective seed dispersal (5), encroachment of riparian vegetation and simplification of river channel habitats (5)
	Reduction in sediment load and resulting downstream channel incision and bed armoring	Reduction in habitat complexity and species richness (5)
	Reduction in longitudinal connectivity	Barrier to migratory fish (1), increased predation of juvenile migratory species (1)
	Conversion of lentic to lotic waters	Elimination of salmonids and native pelagic spawning fishes (1), loss of fish populations from inundation of spawning habitats (1), establishment of exotic species (1,5)
Dam operations	Reduction in flow variability	Increase in exotic fish species (1,4), reduced abundance of fluvial specialists (3)
	Increase in low/base flow magnitude and duration	Reduction in fish populations (1), proliferation of nuisance species (1), physiological stress to riparian vegetation and reduced plant species diversity (5)
	Increased rates of water level fluctuation and erratic flow patterns	Reduction of richness and standing crop of benthic macroinvertebrates (1), washout or stranding of aquatic species (5), decreased macrophyte growth rates and seedling survival (5)
	Loss or shift in timing of seasonal flow peaks	Excessive growth of macrophytes (1), reduction of riparian tree seedling recruitment (2,5), loss of cues for fish migrations and impairment of fish spawning and egg hatching (1,4,5), modification to food web structure (5)
	Modified temperature regimes downstream	Delayed spawning in fish (1), disrupted insect emergence patterns (1), elimination of temperature-limited species of fishes (1), reduced abundance of fish and aquatic macroinvertebrate communities (3)
Interbasin Water Transfers	Increased hydrologic connectivity across natural geographic barriers	Spread of disease vectors (1), translocation of aquatic species outside of natural range (1)

References are limited to reviews and meta-studies in which specific examples of ecological responses to flow regime alterations from water management activities are documented. Sources: (1) Bunn and Arthington 2002; (2) Gasith and Resh 1999; (3) Haxton and Findlay 2008; (4) Murchie et al. 2008; (5) Poff et al. 1997.

Table 3.2 Characteristics of potential pond storage locations (A – N) and priority based on benefit-to-impact ratio.

Pond	Pond Capacity (10^3 m^3)	Storage Benefit (Percent of Total Catchment Water Storage)	Storage Impact (Impact Index, km-day)	Benefit-to-Impact Ratio (Weighted ratio, 1000 x percent/km-day)	Pond Priority
A	33.2	6.5	11.5	5.7	1
B	10.8	2.1	6.2	3.4	2
C	32.6	6.4	46.5	1.4	8
D	13.9	2.7	22.2	1.2	11
E	12.8	2.5	8.1	3.1	3
F	17.1	3.4	16.1	2.1	7
G	108.5	21.4	90.8	2.4	6
H	134.2	26.4	104.7	2.5	5
I	59.5	11.7	39.7	3.0	4
J	17.2	3.4	51.2	0.7	14
K	14.1	2.8	33.4	0.8	13
L	23.4	4.6	41.8	1.1	12
M	13.5	2.7	19.6	1.4	10
N	16.9	3.3	24.2	1.4	9

Figure 3.1 Location of three hypothetical agricultural storage ponds (blue circles) placed in a sub-catchment of the Russian River in northern California, U.S.A. Graphs in right panel show impact of each pond as percent of flow impaired with increasing downstream distance from the pond location.

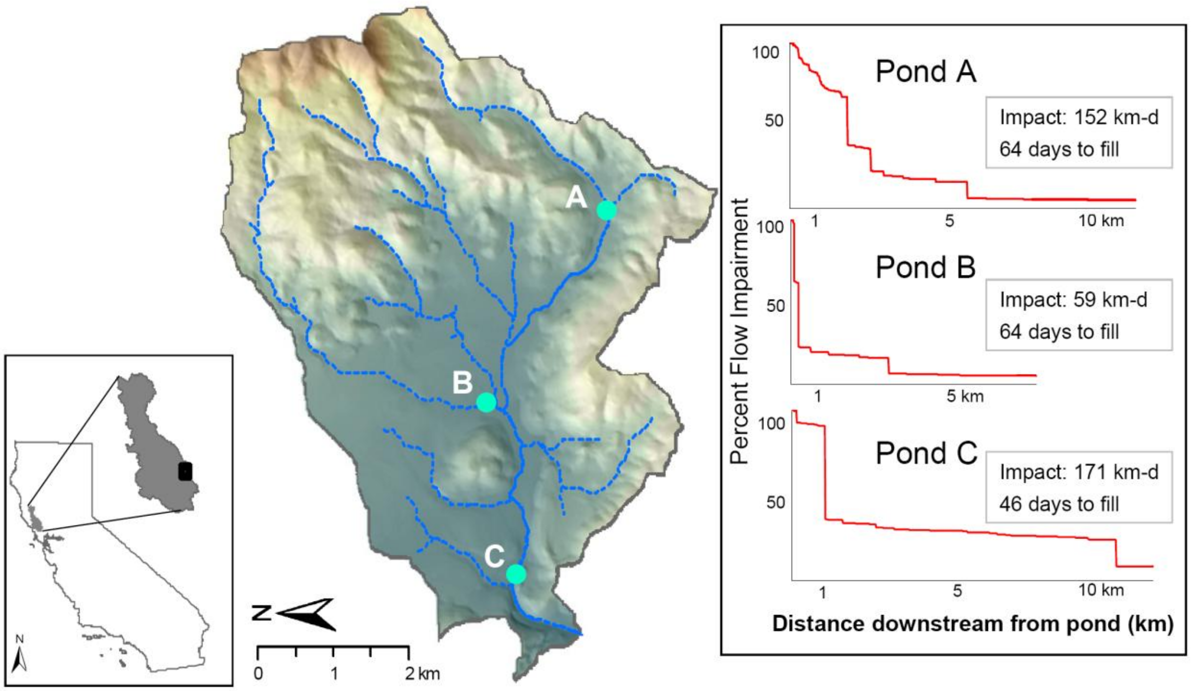


Figure 3.2 Location of storage ponds required meet vineyard water demands on privately owned parcels within the study catchment.

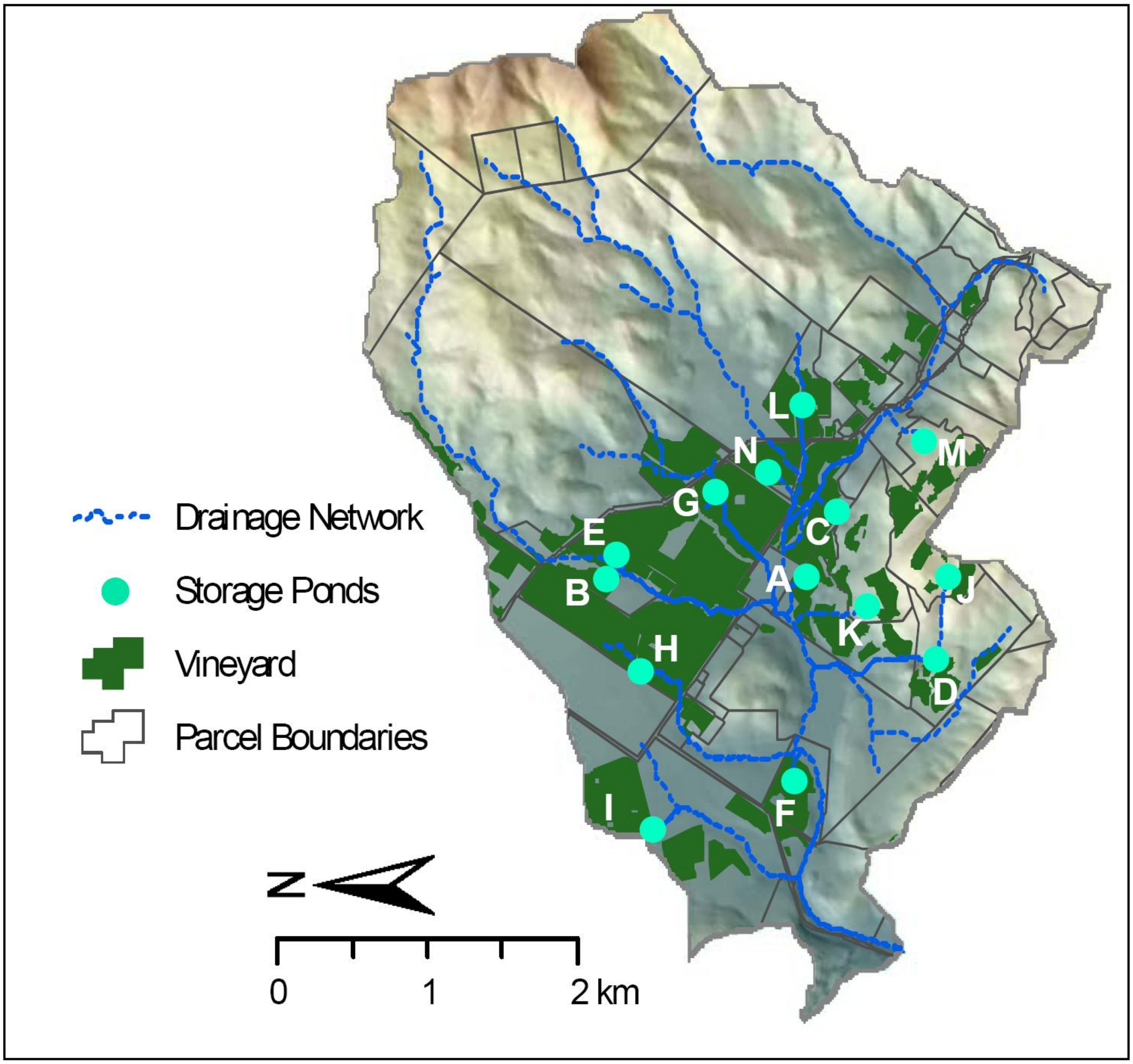


Figure 3.3 Cumulative water storage capacity in catchment by pond priority, based on their benefit-to-impact ratio. Dashed line indicates that approximately 70% of total basin water demand could be met by installing the 6 highest-priority storage ponds.

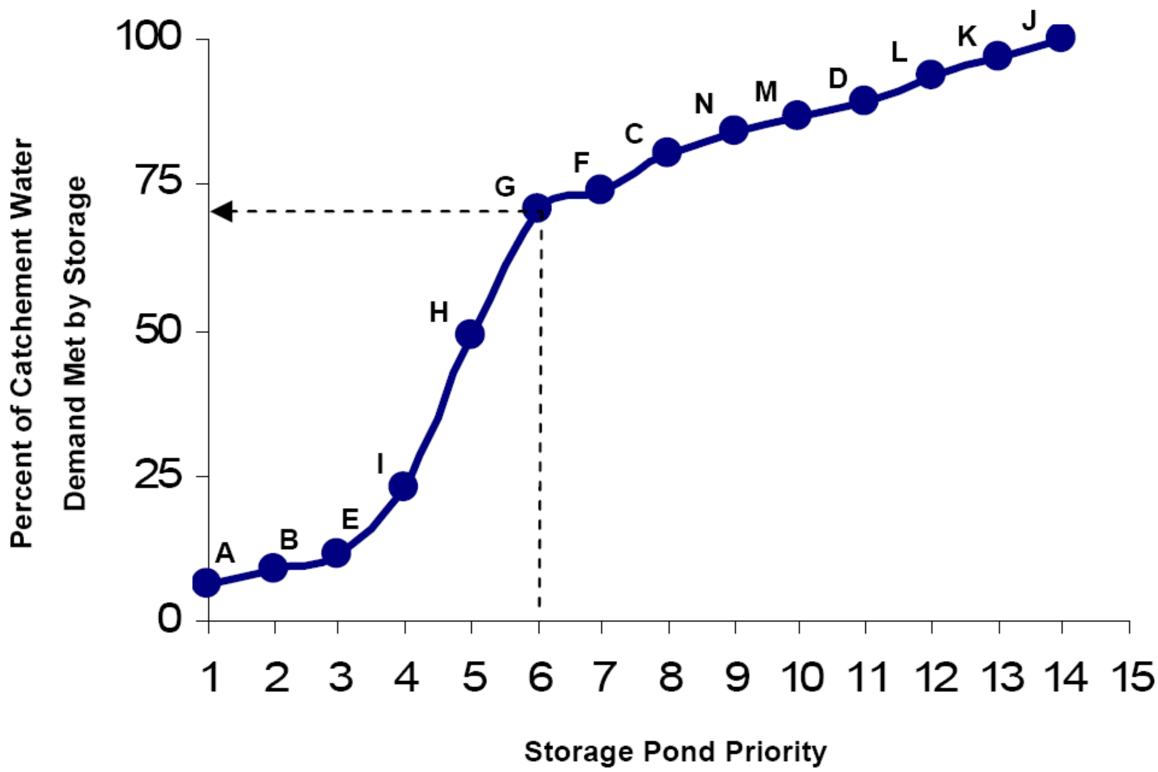
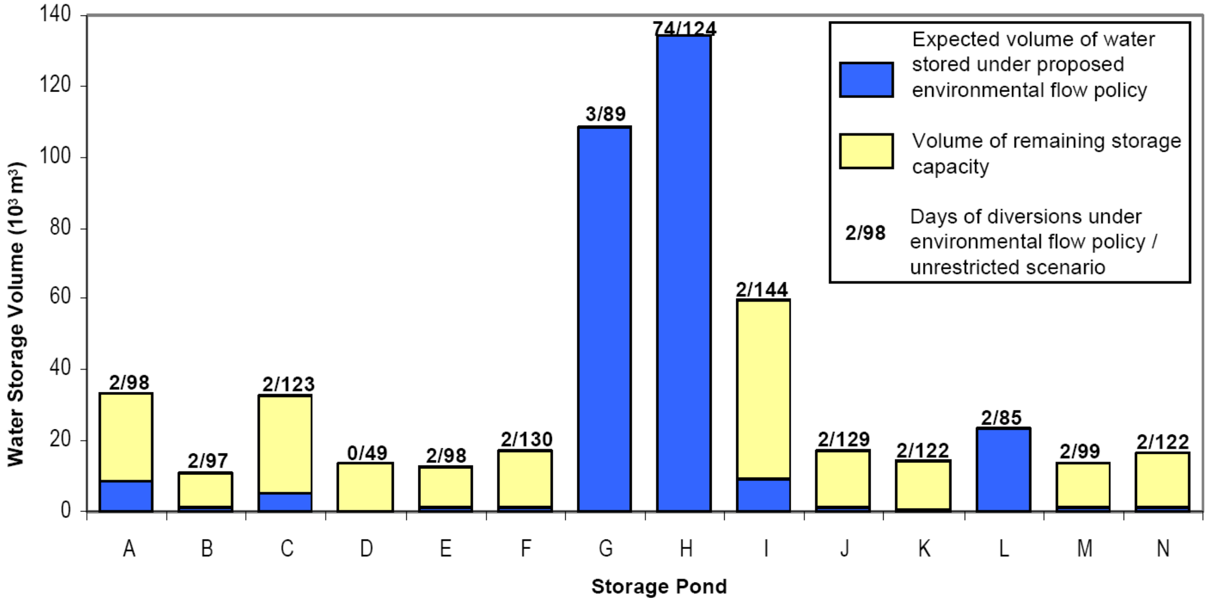


Figure 3.4 Volumes of water stored under an environmental flow policy that restricts period of diversions to allow for fish migrations. Numbers above bars indicate the number of days that water is diverted from streams to fill ponds under the environmental flow policy and when diversions are unregulated. When winter flow diversions are not restricted, all ponds are filled to capacity.



Chapter 4

LiDAR-based hydraulic modeling for salmon passage flow assessments in northern California streams

Theodore E. Grantham

LiDAR-based hydraulic modeling for salmon passage flow assessments in northern California streams

ABSTRACT

The fragmentation of river networks from dams and large diversions is implicated in the decline of Pacific salmon in the western United States and maintaining access to suitable freshwater habitats has become critical to population recovery. Environmental flow standards to protect fish passage are increasingly incorporated in dam operations and have recently been established for streams subject to small-scale diversions. This study introduces a two-dimensional hydraulic modeling approach for evaluating passage flow requirements of steelhead trout (*Oncorhynchus mykiss*) in north coast California streams. High-resolution channel topography data were collected by terrestrial LiDAR surveys and linked with flow velocity and water surface measurements to model stream hydraulics at scales relevant to individual fish. The hydraulic model simulations estimated fish passage flow needs that were substantially greater than those derived by an alternative assessment approach based on water depths at riffle-crest transects, and indicated that shallow-water constraints often occur in the middle of riffles. A regional regression formula used by the State of California to estimate passage flow requirements tended to over-estimate fish passage needs in comparison to the hydraulic modeling approach and preliminary findings suggest the regional formula could be improved by explicitly incorporating a variable describing channel typology. Measured flow data from the 2008-2009 winter season and simulated long-term records indicated that suitable passage flows occur with relatively low frequency and duration (13 to 25 days in most years). Thus, instream flow standards to regulate diversions in small streams appear warranted, though they should be flexible to account for high interannual variation in passage flow days. The hydraulic modeling approach employed in this study is potentially useful for evaluating flow-habitat relationships in stream reaches of particular importance, testing alternative environmental flow assessment methods, and investigating ecological responses to flow regime alterations.

INTRODUCTION

The development of large water projects is implicated in the decline of Pacific salmon populations throughout the western United States. Dams and diversions have extensively fragmented salmon-bearing rivers and streams, which have impacted migration corridors and limited access to historically-utilized habitats (Nehlsen et al. 1991; Good et al. 2005). Flow management from dams has further contributed to the degradation of downstream spawning and rearing habitats (Ligon et al. 1995; Gregory and Bisson 1997). The freshwater life history stages of Pacific salmon are highly adapted to natural seasonal variability in river flows and may be particularly vulnerable to changes in flow regimes caused by water projects (Enders et al. 2009). Therefore, restoring and maintaining access to remaining areas of suitable habitat through environmental flow protections has become an important focus of population recovery efforts for threatened salmon (CDFG-NMFS 2002; ISP 2002).

Environmental flow standards for Pacific salmon have been incorporated into dam operations for many large regulated rivers and are increasingly considered in the regulation and management of smaller water projects. Although small diversions are not capable of altering stream flows to the same extent as large dams, the cumulative effects of several projects within a common stream network can impair ecologically-relevant flows (Merenlender et al. 2008). For example, in the northcoast region of California, streams are generally free of large dams that present direct physical barriers, but fish passage and spawning may be limited during winter low-flow periods due to insufficient water depths at various points along the channel (Vadas 2000). Flows in these streams are often subject to diversions from multiple, small-scale water projects operated by private land owners to meet residential and agricultural water needs (Deitch et al. 2009). Such diversions have the potential reduce frequency and duration of suitable passage flow conditions, already restricted by natural climatic variability.

In response to concerns over the potential cumulative impacts of small-scale water diversions, the California State Water Resources Control Board (SWRCB) adopted the North Coast Instream Flow Policy that prescribes protective measures for fish passage flows by restricting the periods in which water users can divert and store water (SWRCB 2010). The instream flow protection criteria are based on an empirically-derived regression formula that describes how minimum passage flow requirements vary as a function of drainage area and mean annual precipitation. The regression formula was developed from site-specific assessments of fish passage flow requirements in streams throughout the policy region, which involved monitoring water depths at channel transects across a range of flows. Although the conditions for successful passage vary by species and size of individual fish, there is general agreement that upstream movement of adult salmonids may be impaired when water depths fall below 0.2 – 0.3 m (Evans and Johnston 1980; Powers and Orsborn 1985; Bjornn and Reiser 1991). Therefore, the objective of site-specific fish passage studies is to identify the discharge needed to exceed the minimum required water depths within the shallowest reaches of a stream or river channel. These observations are used to construct stage-discharge relationships, which are then fit with a regression line to predict the flow at which minimum passage depths are exceeded. Monitoring transects are typically placed at riffle crests, which is assumed to be the shallowest section of flowing water within a channel reach and therefore represents the critical constraint to fish passage. When the focus of the study is on a single reach, a passage flow threshold is determined by evaluating the minimum flow needed to exceed the passage depth requirement at all transects.

Hydraulic-habitat modeling has also been employed to evaluate site-specific environment flow needs for salmon. In contrast to the riffle-crest approach, which relies on empirically-derived depth-discharge relationships at channel cross sections, numerical hydraulic models compute water depths and velocities based on hydraulic principles, the channel form, and measured or calculated values of stage and velocity at downstream cross sections. In the simplest form, one-dimensional (1-D) hydraulic models predict stage and average velocity at a series of cross sections within a river channel. Flow is assumed to be uniform, with velocities varying only in the longitudinal direction. As a consequence, 1-D model solutions fail to account for transverse variation in flow velocity and depth, such as occur in meander bends, eddies, and recirculation features and which are likely to have biological significance (Ghanem et al. 1996). Although 1-D hydraulic modeling approaches have been widely used to identify environmental flow needs and evaluate the effects of flow regime modifications on fish and aquatic macroinvertebrates (e.g.

Orth and Maugan 1982; Gore et al. 1998, Gallagher and Gard 1999), they are not particularly well-suited for environmental flow studies, where accurately representing the spatial variation in hydraulic conditions at ecologically-relevant scales is critical (Stewart et al. 2005; Souchon et al. 2008).

Fortunately, many of the limitations associated with 1-D models can be overcome with two-dimensional (2-D) models, which are capable of simulating the complex spatial variation of flow fields that occur in natural streams (Leclerc et al. 1995; Ghanem et al. 1996; Crowder and Diplas 2000). In contrast to the cross-sectional description of stream channels used in both the empirical riffle-crest approach and 1-D models, 2-D models rely on a full three-dimensional representation of channel topography, typically defined by a rectangular grid or triangulated irregular network (TIN). Theoretically, 2-D models are capable of reproducing flow features at extremely fine spatial scales, but in practice the size of grid cells used to model hydraulic parameters is often limited by the resolution of available channel topography data. Measuring fine-scale (< 1m) channel features is time-intensive using traditional survey methods (e.g. total station) and as a consequence, most studies that have focused on small-scale hydraulics in relation to aquatic habitat parameters have been restricted to relatively short, single river reaches (Crowder and Diplas 2000; Waddle 2010).

Here, I introduce a 2-D hydrodynamic modeling approach to simulate flows in natural stream channels and evaluate the passage flow requirements of steelhead trout (*Oncorhynchus mykiss*) in north coast California streams. The approach relies on terrestrial LiDAR (LIght Detection And Ranging) surveys to generate high-resolution measurements of stream channel morphology. Terrestrial LiDAR surveys require substantially less effort than traditional survey methods to map meso-scale channel features, which are needed accurately simulate hydraulic conditions at scales relevant to adult salmonids. Although the passage flow requirements of salmonids have been extensively studied, most research has focused on the evaluation of barriers to fish passage at engineered structures such as dams, weirs, and culverts (Thompson 1970; Winter and van Densen 2001; Clarkin et al. 2005). Less attention has been given to fish passage restrictions in stream channels due to natural shallow-water barriers, which can occur at cascades (Reiser et al. 2006) or within riffles (Mosley 1982; Reinfelds et al. 2010). Furthermore, previous environmental flow studies that have used 2-D hydraulic models have to evaluate habitat-discharge relationships have focused on relatively large rivers regulated by upstream dams (e.g. Stewart et al. 2005; Reinfelds et al. 2010), and not on naturally-flowing streams affected by small-scale water diversions. In the present study, field measurements of stream flow, stage and channel substrate were collected to calibrate model simulations of water depth and velocity distributions as a function of discharge in three stream reaches with distinct morphology and catchment size. A fine-scale cell size (0.2 m) was used in the model grid specification in order to simulate water depth distributions at scales relevant to individual fish and evaluate the minimum flow required to maintain a continuous path of suitable passage depths. Fish passage flow requirements indicated by the 2-D modeling approach were then compared with alternative passage flow assessment methods; first by calculating the regional passage flow criteria at each of the study sites and then identifying the minimum flow required to exceed passage depths at riffle-crest transects within the study reaches. Finally, to illustrate the importance of temporal variability in assessing instream flow requirements, I analyze measured and modeled historic discharge records for the study sites to estimate the duration and frequency of passage flows across the range of natural flow variability. The specific objectives of the study were to (i) demonstrate the application of LiDAR-based two-dimensional hydraulic models in fish passage flow studies, (ii) use model simulation results to evaluate the outcomes and assumptions of alternative passage flow assessment methods, including the regional formula and riffle-crest approaches, and (iii) examine the influence of inter-annual flow variability on the frequency and duration of suitable passage flow conditions.

METHODS

Study Area

Gill Creek and Sausal Creek are tributaries to the Russian River in Sonoma County, California (Figure 4.1). The Gill Creek watershed (16 km^2) is located approximately 3-km north of the Sausal Creek watershed (33 km^2). Mean daily discharge is approximately 0.32 and 0.61 m^3s^{-1} for Gill and Sausal Creek, respectively. Both streams originate on the southwest-facing slope of the Mayacamas Mountains and descend from approximately 875 to 50 m above sea level through steep, confined channels characterized by large cobbles, boulder and bedrock substrate. The streams then pass through the alluvial plain of Alexander Valley to their confluence with the Russian River. In the alluvial zone, stream channel widths increase and the bed is comprised primarily of gravel and sand sediments. The steep upper portions of the catchments are covered by native oak woodland vegetation and hillslope vineyards. The lower portions of the catchments are dominated by vineyard agriculture.

The region is characterized by a Mediterranean climate, with mean annual rainfall of approximately 800 mm. Nearly all annual rainfall occurs between November and May, and is typically delivered in brief, intense storms. Winter stream flow tracks rainfall patterns yielding a flashy hydrograph with peak flows that are often orders of magnitude greater than baseflows. During the dry season (June to October), flows in these streams gradually recede to intermittent conditions. In the confined, upper channel reaches streams contract to a series of pools while in the lower, alluvial reaches the stream channel completely dries by mid-July.

Terrestrial LiDAR topographic surveys and data processing

Three study reaches, each comprising approximately 200 m of stream length, were selected to investigate the relationships between stream flow, channel morphology, and passage suitability for adult steelhead trout. A confined channel reach and an unconfined, alluvial channel reach were identified on Sausal Creek and a single confined channel reach was identified on Gill Creek (Figures 4.1 and 4.2; Table 4.1). The study reaches were at least 20 channel widths in length, which is considered an appropriate scale over which to characterize local stream channel processes and habitat characteristics (Montgomery and Buffington 1997). A terrestrial laser scanner (I-SITE 4400 LR) was used to collect high-resolution topographic data of the stream channel in each of the study reaches. The scanner was mounted on a tripod and positioned with the stream channel in 7 to 10 locations. Neighboring scan positions were separated by less than 40 m to ensure fairly continuous, high-resolution (<0.25 m) coverage of data points within the survey area. Each scanner location was mapped with a total station (Nikon DTM-352) and geo-referenced to established survey control points (e.g. benchmarks). At the time of the surveys (October 2008), the streams in the Gill Creek Canyon and Sausal Creek Canyon sites were flowing at a low rate (<0.001 m^3s^{-1}) and occupied less than 10 percent of the channel. Because the laser does not penetrate water, a total station was used to survey individual points within the wetted areas and map topography beneath the water surface. The stream channel was completely dry at the Sausal Creek Alluvial site when the survey was conducted.

Data processing was performed using the I-SITE Studio software program (Maptek 2010), which is designed to visualize and manipulate large point files of terrestrial laser scanning data. Each laser scan yields a cloud of points reflected off surfaces surrounding the LiDAR scanning unit. The survey data from each scan were merged based on the geographic coordinate location of each scan and orientation to a survey benchmark. A series of topographic filter were then applied to eliminate survey points outside of the study area, remove points reflected off vegetation above the ground surface, and to limit the minimum distance between points to 0.1 m. Finally, the survey points beneath the water surface were added to complete the surface topography data set. The final survey for the Gill Creek, Sausal Canyon, and Sausal Alluvial reaches included 88 006 X,Y,Z coordinate points, 232 898 points, and 182 690 points, respectively. Point densities were similar among all sites and differences in the total number of points reflect variation in the area surveyed for each site.

Flow and hydraulic habitat monitoring

To measure changes in flows within the study reaches, pressure-transducer type water level gages (In Situ) were installed to record stage at 10-minute intervals for the 2009 water year (October 1, 2008 to September 30, 2009). Each pressure transducer probe was encased in flexible PVC tubing and attached to solid substrate at the bottom of a pool within the stream channel. Measurements of water surface elevation were transmitted to a logger, where the data were retrieved by manually downloading the data to a PC. Discharge measurements were taken at approximately bi-weekly intervals using Pygmy or AA current meters. Velocities were measured at 10-15 points along a cross-section at a location of the stream channel with approximately uniform flow. In water less than 0.8 m deep, measurements were taken at 0.6 depths to estimate mean column velocity. At greater than 0.8 m depths, velocity measurements were taken at 0.8 and 0.2 m depths and the recorded values averaged. The point velocities were multiplied by the estimate area of each vertical column of the cross-section and then summed to obtain the discharge. The discharge measurements and continuous stage data were then used to develop rating curves and generate a continuous record of flow conditions at the site, in accordance with standard USGS methods for stream gaging (Rantz et al. 1982).

Water surface elevations were surveyed at three to five discharge levels between November 2008 and May 2009 using a total station (Topcon GTS-213) and prism reflector. Coordinate point locations were recorded at the water's edge on both sides of the stream channel along the full extent of the study reach. Measurements were taken at all habitat transitions (e.g., pool to riffle) along the reach to capture changes in water surface slope. During one of the visits to each site, water depth and velocity measurements were also taken at 2-3 representative cross-sections and 30-50 randomly selected points within the stream channel. A current meter was used to measure the average velocity of the vertical water column. The location of each velocity measurement was surveyed with the total station and prism reflector.

The channel substrate in each reach was characterized by mapping regions of distinct sediment sizes, based on the length of the intermediate axis of the dominate particle size (Dunne and

Leopold 1978). Patches of similar sediment-sizes were visually mapped onto an image of the surface topography generated from the LiDAR survey data. Sediment patches were classified as sand (less than 0.0025 m), gravel (0.05 m), gravel/cobble (0.08 m) and large cobble (0.15 m), and boulder (0.2 m). Patches vegetated with willow (*Salix* spp.) within the stream channel were also mapped and assigned a sediment size value of 0.3 m, to account for the influence of vegetation on channel roughness (Wu et al. 1999).

Hydraulic modeling

Topography, flow, and water surface elevation data were used as input to the Multi-dimensional Surface Water Modeling System (MD_SWMS), a graphical user interface developed by the U.S. Geological Survey for numerical models of surface-water hydraulics and sediment transport in rivers (McDonald et al. 2005). The topographic data were converted into triangulated irregular network (TIN) terrain models and mapped onto a curvilinear orthogonal coordinate 0.2-m grid oriented along the centerline of the channel. MD_SWMS was used to run a depth-averaged two-dimensional flow model, Flow and Sediment Transport and Morphological Evolution of CHannels (FaSTMECH) to simulate hydraulic conditions at different discharges (McDonald et al. 2005). FaSTMECH is designed to model flow hydraulics in natural rivers and streams and calculates both downstream and transverse velocity components and flow depths at each node in the model grid. The model runs iteratively using a finite-difference scheme to solve for the momentum equations and water surface elevation at each node until it converges on a steady-state solution.

A down-stream water surface elevation and discharge must be specified as boundary conditions to run the model, which is then calibrated by adjusting a uniform drag coefficient parameter until the predicted and measured water-surface elevations are in agreement. Patches of variable roughness are then specified and run in the model using the sediment size data. Drag coefficient values are estimated at each node from the mapped dominant grain size and the depth solution from the simulation using a constant drag coefficient (McDonald et al. 2005). The drag coefficient (C_d) calculation assumes a logarithmic velocity profile and is based on the following equation:

$$C_d = \left[\frac{1}{k}\left(\ln\left(\frac{h}{z_0}\right) - 1\right)\right]^{-2} \qquad [1]$$

where is h is the depth of flow, k is Von Kármán's constant equal to 0.403 and z_0 the roughness length parameter, which is estimated as the grain size times a user-defined constant, set at 0.15 for all simulations. In addition, the model user must define a lateral eddy viscosity to represent lateral momentum exchange due to turbulence or other variability not generated at the bed (Nelson et al. 2003). In FaSTMECH, the lateral eddy viscosity is computed and iteratively applied during the calibration simulations using the equation:

$$LEV = 0.01 * u_{avg} * d_{avg} \qquad [2]$$

where LEV is the lateral eddy-viscosity coefficient (m^2s^{-1}), u_{avg} is the average velocity (ms^{-1}) and d_{avg} is the average depth (m) from all nodes in the solution grid. The computed LEV value was applied uniformly through the modeling reach for each calibration stream flow.

The two-dimensional hydraulic model was first used to run steady-state flow simulations at discharges for which water surface elevations were measured. Based on the relationship between measured downstream water surface elevation and the recorded stage at the flow gage, a linear interpolation was used to predict the downstream water surface elevation across a range of discharges between 0.1 and 5 m^3s^{-1}. At each site, a total of 12 flow simulations were performed to generate predictions of depth and velocity for all wetted nodes in the model grid.

Passage flow assessment

To assess how changes in discharge and stream hydraulics could affect adult steelhead trout migrating through the study reaches, I quantified the extent of suitable passage depths within the channel over the range of simulated discharges. Minimum passage depth required for adult fish passage was defined as 0.25 m, consistent with current instream flow policy criteria for steelhead trout (SWRCB 2010). This depth criterion is based on previous studies of passage suitability criteria (e.g., Thompson 1970; Powers and Orsborn 1985) and is related to the body depth of a typical adult steelhead trout (0.15-0.20 m) swimming 0.05 m above the streambed. An individual fish moving through the channel is assumed follow a continuous path, from the downstream to upstream end of the reach. Therefore, the most accessible route for a fish would follow a least cost path, where shallower waters are avoided and deeper waters preferred. To identify the least cost migration path, the solution grid of simulated depths was imported from MD_SWMS into an ArcGIS (ESRI 2009) spatial data file. The inverse depth of each node was specified as the cost layer and the least cost path was computed using the Spatial Analysis Extension (McKoy and Johnston 2001) to identify the potential migration route along the deepest, contiguous path of cells, extending from the downstream to the upstream end of the reach. A series of adjacent cells along the route that exceed the minimum depth threshold comprise a ‘suitable’ path segment. The total length and number of suitable path segments that exceed the minimum depth threshold increases with increasing flow, until complete passage flow connectivity is achieved and all path segments meet or exceed the minimum depth threshold.

Several metrics were calculated to describe the extent of suitable passage depths along the migration path, including the proportion of cells exceeding the depth criterion, the total path length of suitable depths, and the number and length of unsuitable (shallow) path segments. Only path segments of two or more contiguous cells (equal or greater than 0.4 m in length) were considered in the analysis. To quantify the sensitivity of the discharge-passage suitability relationships to the defined passage criterion at 0.25 m, I also calculated each of the path metrics for minimum passage depths of 0.15, 0.2, and 0.3 m.

The influence of velocity on passage suitability was not considered in this analysis. Based on the characteristics of the stream channels in the study reaches and the range of flows considered in this analysis, it is unlikely that fish passage would be limited by velocity barriers. In general, high water velocities become an effective barrier when the entire flow becomes concentrated in a fast chute, the length and speed of which combine to overcome the fish’s swimming ability. These conditions are most often encountered at culverts or natural cascades (Reiser et al. 2006), which do not occur in the study reaches. Furthermore, previous studies indicate that steelhead trout are capable of sustained swimming against currents of 3 ms^{-1}, and can achieving burst swimming speeds of 4 – 8 ms^{-1} to negotiate falls and high-velocity areas (Stringham 1924;

Powers and Orsborn 1985), and thus are unlikely to be impaired by flow velocities encountered in the study reaches over the range of simulated discharge.

Comparison to alternative passage flow assessment methods

To evaluate the results of the 2-D modeling passage flow analysis in the context of alternative instream flow assessment methods, I compared site-specific minimum passage flow estimates to those protected under the current policy framework for the region. The California State Water Board Instream Flow Policy (SWRCB 2010) focuses on maintaining natural flow patterns in northern California rivers and streams to protect anadromous salmonid populations. The policy relies upon a regional formula, which defines minimum bypass flow (Q_{mbf}) as the "minimum instantaneous flow rate of water that is adequate for fish spawning, rearing and passage as measured at a particular point in the stream," and is determined by a regionally-derived formula

$$Q_{mbf} = 8.8\, Q_m (DA)^{-0.47} \qquad [3]$$

where Q_m is the mean annual unimpaired flow in cubic feet per second and DA is the catchment drainage area in square miles. The mean annual unimpaired flow (Q_m) is typically estimated at the point of interest from long-term flow records at the closest gage station, adjusted by drainage area and mean annual precipitation.

The Policy also stipulates that as an alternative to the regional formula approach (hereafter, 'regional approach'), site-specific studies may be conducted to determine minimum passage flow requirements. A recommended approach for assessing passage flows at the site level involves identifying depth-discharge relationships at riffle crests (hereafter, 'riffle-crest approach'). The riffle crest is an area of accelerating flow associated with a distinct increase in water surface slope, where the stream channel transitions from a deeper area of slow-moving water (e.g., pool) to a shallower area of rapidly flowing water. The shallowest flows within a natural stream channel are often at the riffle crest, which is why it is often used as a reference to identify minimum flow requirements for fish passage. The assessment method involves monitoring water depths at several riffle crests across the range of characteristic discharges and determining the flow needed to maintain depths sufficient to allow fish passage. To compare this approach with the two-dimensional hydraulic modeling method employed in this study, cross sections were extracted from the flow solutions at every riffle crest within the modeled reaches. Depths at the riffle crest thalweg were plotted against discharge for each cross-section and the minimum flow necessary to maintain 0.25 m depths was estimated from the flow-discharge curve.

Temporal variability of passage flows

The development of minimum passage flow recommendations requires not only consideration of spatial hydraulic patterns (e.g., distribution of depths as a function of discharge), but also the temporal dynamics of stream flow. The study reaches, located in a Mediterranean-climate region, are characterized by highly variable seasonal and inter-annual hydrograph that influences the periods of potential fish passage. To evaluate the temporal variability in minimum passage flows at the study sites, I first examined the site-specific discharge data collected at 10-minute intervals for the 2008-2009 steelhead migration period. For each site, I used the minimum flow required for fish passage obtained from the 2-D modeling solution to calculate the frequency and duration the flow threshold was exceeded between November 1 and March 31. I next examined interannual variation in passage flow patterns by simulating 20 years of daily flows records at the study sites, based on gaging records from Maacama Creek (#11463900; 1961-1980) located about 5 km south of Sausal Creek (Figure 4.1), using daily flow values scaled by drainage area and mean annual precipitation. Passage flow duration and frequency statistics were then calculated for each year of the simulated record.

RESULTS

Hydraulic model solutions

The water surface elevation predictions from the variable-roughness simulations were compared to measured values not used during model calibration in order to quantify model error (Table 4.2). The predicted water surface elevations and depths were generally within 0.1 m of the measured values for Gill Canyon and Sausal Canyon and within 0.2 m of measured values in Sausal Alluvial canyon (Figure 4.3). Predicted point velocities were generally within 0.3 ms^{-1} of measured values at all sites. At simulated flows of 0.6 m^3s^{-1}, modeled depths at Gill Canyon were between -0.07 and 0.12 m of measured values, with a mean residual error of 0.036 m (Figure 4.4A). At the same flow rate, Sausal Canyon model depth errors were between -0.20 and 0.27 m, with a mean residual error of 0.10 m, and Sausal Alluvia depth errors were between -0.12 and 0.2 m with a mean residual error of 0.07 m. Mean residual error for predicted velocities at 0.6cms was 0.15 ms^{-1} for Gill Canyon, 0.14 ms^{-1} for Sausal Canyon, 0.19 ms^{-1} for Sausal Alluvial (Figure 4.4B).

The hydraulic model simulation results describe how changes in flow interact with local channel morphology to produce variable patterns in velocity and depth distributions. Increasing discharge is associated with a shift in depth and velocity distributions to higher values for all sites (Figures 4.5 and 4.6). However, there are reach-specific responses to flow that reflect differences in channel morphology. In the Gill and Sausal Canyon reaches, the confined stream channels resulted in depth distributions that were restricted to a more narrow range of values than the Sausal Alluvial reach, which has a relatively broader channel such that increasing flows tend to extend laterally, thus maintaining relatively low water depths (Figure 4.5). The shifts in velocity distributions followed the same general pattern, but also revealed unique differences among reaches. The relative increase in velocities with increasing discharge was greater for the confined, canyon reaches in comparison to the alluvial reach. Between the canyon reaches, Gill Canyon tended to have higher velocities than the Sausal Canyon reach across the range of discharges.

Minimum passage flows

The predicted migration route, as defined by the deepest path of contiguous cells from the bottom to the top of each study reach, was used to assess passage suitability across the range of simulated flows. With increasing discharge, the model solutions indicate an increase in the proportion of path segments at or above the 0.25-m depth threshold (Figure 4.7) and an expansion of the wetted channel area (Figures 4.8 – 10). At low flows (e.g. Figure 4.8A), segments of the channel that fall below the minimum depth threshold predominately occur in riffles. The water depths in the riffles increase with flow, expanding the areas of suitable passage depths and increasing connectivity along the stream channel. The proportion of nodes along the migration route at or above the depth threshold also increase with discharge and the length of gaps below the depth threshold decrease (Figure 4.7). The hydraulic model simulations for Gill Canyon indicate that a discharge of 1.30 m^3s^{-1} is required to provide complete connectivity of suitable passage depths along the entire migration path. The discharge required to provide full passage depth connectivity is approximately 1.10 m^3s^{-1} at Sausal Canyon and 0.90 m^3s^{-1} at Sausal Alluvial.

To evaluate how the assessment of minimum passage flows is affected by the depth criterion, I identified the flows required to provide flow connectivity along the migration path at 0.15 m, 0.2 m and 0.3 m depths (Table 4.3). In the canyon reaches, passage flows are more sensitive to changes in passage depth criteria than the alluvial reach. Required passage flows at Gill Canyon range from 0.7 m^3s^{-1} when calculated at the lower, 0.15 m depth threshold, to 2 m^3s^{-1} at the upper, 0.3 m depth threshold. At Sausal Canyon, minimum passage flows range from 0.4 to 1.6 m^3s^{-1} when calculated for the lower and upper depth criteria, respectively. In contrast, passage flows at Sausal Alluvial fell between 0.7 and 1.1 m^3s^{-1} when evaluated for the same range of depth thresholds.

Based on the State's regional approach, minimum passage flows at the study reaches would be 1.24 m^3s^{-1} at Gill Canyon, 1.77 m^3s^{-1} at Sausal Canyon, and 1.76 m^3s^{-1} at Sausal Alluvial. Therefore, the State's approach yields results that are comparable to the site-specific hydraulic modeling results for Gill Canyon, but that are significantly higher than those obtained for Sausal Canyon and Sausal Alluvial (Table 4.4). Furthermore, the regional approach indicates that passage flows at the Sausal Canyon and Alluvial sites are essential identical, while the hydraulic modeling approach suggest that Sausal Canyon requires 20 percent more flow than the Alluvial reach to allow fish passage.

The depth-discharge relationships derived from the hydraulic model solutions at riffle-crest cross sections indicate substantially lower passage requirements than those obtained by both the hydraulic modeling and the regional formula approaches (Table 4.4). At Gill Canyon, a discharge of 0.75 m^3s^{-1} is sufficient to inundate all of the riffle cross sections to water depths 0.25 m. Sausal Canyon requires only 0.25 m^3s^{-1} and Sausal Alluvial 0.5 m^3s^{-1} to meet the passage depth criteria. Therefore, the riffle-crest approach suggests that fish passage is possible at approximately 25 – 50 percent of the discharge specified by the hydraulic modeling approach.

Temporal variability of passage flows

The 2009 water year winter (between Nov. 1, 2008 and Mar. 31, 2009) had relatively low rainfall and only four major storms during the winter season (Figure 4.11). Based on the results derived from the hydraulic modeling analysis, flows at Gill Creek were adequate for fish passage during 5 events (separated by at least 12 hours), each lasting between 40 minutes to 2.2 days (Table 4.5). In total, the estimated duration of passage flows at Gill Canyon was 6.2 days. Estimated passage flows at Sausal Canyon and Sausal Alluvial occurred on 6 events (lasting from 3.5 hours to 6.1 days) and 7 events (lasting from 1.8 hours to 13.7 days), respectively. The total estimated duration of passage was 18.9 days at Sausal Canyon and 22.5 days at Sausal Alluvial. Based on the 20 years of simulated hydrographs for the study reaches, the duration and frequency of passage flows varied substantially among years (Table 4.5). On average, there were 5.4 passage flow events per year at Gill Canyon and 5.9 passage flow events per year at Sausal Alluvial. The median seasonal duration of passage flows at Gill Canyon was 18.5 days, but the total duration within each year ranged from 0 to 43 days. At Sausal Canyon, the median duration of passage flows was 37 days, ranging from 0 to 80 days over the 20-year period of analysis. Seasonal passage flow duration at Sausal Alluvial had a median value of 45 days, ranging from 0 to 101 days.

DISCUSSION

When informed by detailed topographic data of the stream channel and calibrated by measurements of water surface elevation and velocity, 2-D hydraulic models are a powerful tool for assessing fish passage flow requirements in small streams. The 2-D hydraulic modeling approach described in this study relies on a continuous, fine-scale topographic surface of the stream channel and yields accurate estimates of the spatial distribution of water depths and velocities. Because the accuracy of model predictions are limited by the resolution of the channel surveys (Waddle 2010), the utility of 2-D models in environmental flow studies are only likely be realized if detailed bathymetric data is available. The terrestrial LiDAR surveys generated fine-scale (<0.25m) terrain models for the study streams over relatively long reaches, and thus offer a promising alternative to traditional survey methods (e.g., total station) in terms of accuracy, resolution, and efficiency. Furthermore, terrestrial LiDAR is capable of surveying relatively small streams channels covered by riparian canopy cover, which are difficult to survey with aerial LiDAR or other remote sensing techniques used in river studies (e.g., McKean et al. 2008; Cavelli et al. 2008). Because the terrestrial LiDAR scanner cannot penetrate water, the survey method is probably not suitable for large, perennial streams in which a large proportion of the channel lies beneath the water surface.

The discharge required to provide a continuous path of suitable passage depths was influenced by local channel morphology and varied between 0.90 and 1.30 m^3s^{-1} in the three study reaches. While the regional approach for defining passage flow requirements yielded comparable or higher minimum flow values than indicated by the 2-D model simulations, the approach was not sensitive to changes in channel morphology that occur over small distances. For example, predictions for minimum passage flows by the regional method were similar for Sausal Canyon and Sausal Alluvial, which have similar drainage areas despite their distinct differences in channel width, slope, and sediment sizes (Table 4.1; Figure 4.2). One would expect confined, canyon reaches to have narrower channels and thus have relatively deeper flow depths than the broader, alluvial channel reaches at the same flow rate. However, in this example, Sausal Canyon had slightly higher passage flow requirements than the Sausal Alluvial reach. This could be because the Sausal Alluvial reach has been modified to some extent by bank stabilization structures, resulting in channel incision and a more restricted cross-section profile than typical for a gravel-dominated, gently-sloped reach. The passage flow threshold derived from the regional approach is based only on drainage area and mean annual flow. While the simplicity of the regional approach makes it an attractive tool for management and decision-making, potential improvements to the formula could be made by incorporating a variable that accounts for changes in channel morphology associated with geologic transitions (e.g., bedrock-dominated confined to gravel-dominated alluvial reaches) or anthropogenic influences (e.g., channel stabilization).

In comparison to the 2-D modeling method, monitoring water depths at riffle crests consistently underestimated minimum flow requirements because low water depths persisted in sections of the channel even when minimum passage depths were exceeded at riffle crests (Figures 4.8 – 10). Although a comprehensive investigation of stage-discharge relationships at riffle crests would typically use more cross sections (e.g., 15-20) than evaluated in this study, the results suggest that the riffle-crest approach is probably not adequate for assessing passage flow requirements. Where the use of 2-D hydraulic models to assess site-specific passage flows is not feasible, the findings suggest that passage depth monitoring should include transects within long riffles. While it may be difficult to predict the location of the shallow water constraints within riffles a priori, including a higher density of monitoring transects is likely to yield results closer to the passage flow connectivity threshold derived from the 2-D modeling approach.

The results also highlight the importance of interannual flow variability in assessing instream flow needs. Based on the calculated passage requirements for each reach, passage flows occurred fewer than 7 times at all sites during the 2009 winter season. Furthermore, the long-term hydrographs modeled for each site indicate that the total duration of passage flows ranges from 13 to 25 days in most years. Because passage flows occur with relatively low frequency and duration, establishing standards to regulate diversion operations appear warranted. The large variation in passage flow days also suggests that flexibility should be incorporated in environmental flow regulations. Because the ecological consequences of temporarily impairing passage likely to be more significant in years with few passage flow days than in years with many opportunities for fish passage, regulations should be more restrictive in dry than in wet years. It is also important to recognize that the more protective regulations are of passage flows, the less frequently flows will exceed the minimum threshold, thus providing fewer opportunities for water users to meet their storage demands. This could have indirect environmental consequences, such as shifting water-use pressures to the dry season and intensifying the adverse effects associated with low-flows on juvenile fish populations (Grantham et al. 2010). Regardless of the specific flow criteria selected, an awareness of the seasonal and interannual dynamics of passage flows is critical to designing effective management strategies (Stalnacker et al. 1996).

The importance of hydrologic connectivity for maintaining river ecosystem integrity is well-documented (Pringle 2003; Freeman et al. 2007). Therefore, the method for evaluating passage flow connectivity along the channel migration path may be a useful tool in other environmental studies that require quantitative metrics describing hydrologic connectivity as a function of discharge. Because this study was focused on depth constraints to fish passage, field measurements and model calibration was focused on minimizing error in water surface elevations. An investigation of velocity distributions or other hydraulic parameters relevant to ecological preferences would require additional field measurements for adequate model calibration but could be supported in the same modeling framework.

The effort required to initialize and calibrate flow simulations also places practical constraints on the spatial extent to which 2-D modeling approach can be applied. Managers responsible for regulating water rights and protecting fish and wildlife generally prefer tools that are less resource intensive and that can be readily extrapolated to other sites across spatial scales. Nevertheless, 2-D models are capable of producing detailed and accurate representations of hydraulic habitat conditions that are difficult to generate from alternative approaches. The spatially-explicit simulation of flow depths and velocities produced by hydraulic model is also useful for visualizing and communicating the consequences of flow regime alterations on stream hydraulics and habitat suitability. Therefore, the 2-D hydraulic modeling approach may be particularly valuable for testing the assumptions behind alternative environmental-flow assessment methods or for evaluating site-specific habitat-flow relationships in reaches of particular importance (e.g. critical habitat for threatened species or sites subject to potential alterations from water or land use development).

Uncertainty in the relationships between river flows and biological responses is a persistent challenge that must be dealt with in all environmental flow assessments (Castleberry et al 1996; Railsback 1999; Poff and Zimmerman 2010). In this study, fish passage is assumed to be constrained by segments of shallow water along the migration path. This is clearly a simplification of fish behavior because adult salmon have been observed working their way upstream over shallow riffles with a significant proportion of their bodies exposed above the water surface. Thus, the hydraulic factors that influence the probability of fish passage not only relates to local depths, but also to the lengths of shallow water flow that occur within the stream channel. Furthermore, changes in the minimum depth criteria substantially influenced the recommended passage flow, suggesting that the body size and swimming ability of an individual fish will be important for determining the likelihood of successful passage. Linking calibrated hydraulic model predictions with detailed observations of fish passage would be highly informative for quantifying how flow depths and lengths of shallow-water reaches interact to influence fish passage behavior. The spatially-explicit characterization of stream habitat is critical for establishing mechanistic ecological-flow relationships (Souchon et al. 2008). Therefore, high resolution 2-D hydraulic modeling offers a promising means of linking physical environmental controls to ecological responses and advancing research and understanding at the biological-physical interface.

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Table 4.1 Description of study reaches

Site	Catchment area (km^2)	Reach length (m)	Mean slope (mm^{-1})	Bankfull width (m)[2]	Median winter discharge (cms)[1]	Channel morphology
Gill Canyon	14.6	200	0.012	4.3	0.20	confined
Sausal Canyon	27.4	210	0.013	7.1	0.37	confined
Sausal Alluvial	28.6	195	0.008	7.4	0.38	alluvial fan

[1] Mean daily winter (November 1 to March 31) discharge estimated from 20-years of modeled data from USGS gage station records
[2] Mean width of low-flow channel

Table 4.2 Simulation summary for model calibration

Site	Date	Discharge (m^3s^{-1})[1]	Downstream stage (m)[1]	Lateral eddy viscosity (m^2s^{-1})	Uniform drag coefficient (dimensionless)	WSE mean residual error (m)	WSE residuals range (m)[2]
Gill Canyon	2/12/2008	0.06	87.830	0.0002	0.3	0.053[2]	-0.11 to 0.15
	3/5/2009	0.44	87.930	0.001	0.11	0.035	-0.10 to 0.07
	3/4/2010	0.62	87.948	0.001	0.085	0.038	-0.14 to 0.05
	2/22/2009	14[3]	88.490	0.01	0.07	0.101	-0.16 to 0.21
Sausal Canyon	4/30/2009	0.09	91.900	0.0002	0.18	0.061	-0.15 to 0.2
	2/25/2010	0.61	92.015	0.0008	0.06	0.041	-0.08 to 0.11
	2/20/2009	0.67	92.049	0.0009	0.05	0.042	-0.02 to 0.13
Sausal Alluvial	1/9/2009	0.02	95.100	0.0001	0.25	0.096	-0.1 to 0.13
	2/25/2010	0.65	95.300	0.001	0.09	0.099	-0.05 to 0.23
	2/19/2009	1	95.400	0.0015	0.04	0.114	-0.14 to 0.20

[1] Measured discharge, stage and water surface elevation data used for model calibration
[2] Difference between measured and predicted water surface elevations
[3] Simulation calibrated with high water mark survey elevations and peak discharge estimate

Table 4.3 Sensitivity analysis of minimum passage depth criteria on passage flow needs

	Minimum passage flow (m^3s^{-1})		
Passage Depth Criteria	Gill Canyon	Sausal Canyon	Sausal Alluvial
0.15 m	0.7	0.4	0.7
0.20 m	1.25	0.8	0.75
0.25 m	1.3	1.1	0.9
0.30 m	2	1.6	1.1

Table 4.4 Comparison of passage flow assessment methods

	Gill Canyon		Sausal Canyon		Sausal Alluvial	
Method	Min. Bypass Flow (m^3s^{-1})	% Connectivity	Min. Bypass Flow (m^3s^{-1})	% Connectivity	Min. Bypass Flow (m^3s^{-1})	% Connectivity[2]
$Q_{connectivity}$[1]	1.30	100[2]	1.10	100[2]	0.90	100
$Q_{riffle\ crest}$	0.75	90	0.25	75	0.50	91
$Q_{regional}$	1.24	99	1.77	100	1.76	100

[1] Passage flow estimate from predicted migration path within modeled reach
[2] Percent of cells along predicted migration path that meet or exceed 0.25 m depth.

Table 4.5 Passage flow frequency and duration statistics for measured 2009 discharge and modeled long-term discharge (1961-1981)

Site	Passage flow criteria (m^3s^{-1})	Frequency (# spells)	Mean Duration (days)	SD Duration (days)	Min Duration (hours)	Max Duration (days)	Total Duration (days)
Gill Canyon	1.3	5 (5.4)[1,2]	1.2 (19.6)	1.1 (13.3)	0.7 (0)[3]	2.2 (43)	6.2
Sausal Canyon	1.1	6 (5.9)	3.1 (31.8)	3.1 (21.6)	3.5 (0)	6.1 (80)	18.9
Sausal Alluvial	0.9	7 (5.9)	3.2 (45.0)	5.2 (24.8)	1.8 (0)	13.7 (101)	22.5

[1] Statistics based on 10-minute flow data 2009 and simulated mean daily flow for 20-year record (1961-1981), shown in parentheses.
[2] Spells are defined as instantaneous flow exceeding the minimum passage flow criterion, separated from by at least 12 hours for 2009 records or 1 day for long-term records.
[3] In 1977, the minimum passage flow was not exceeded.

Figure 4.1 Study locations in the Gill Creek and Sausal Creek catchments, along the Russian River in California, USA.

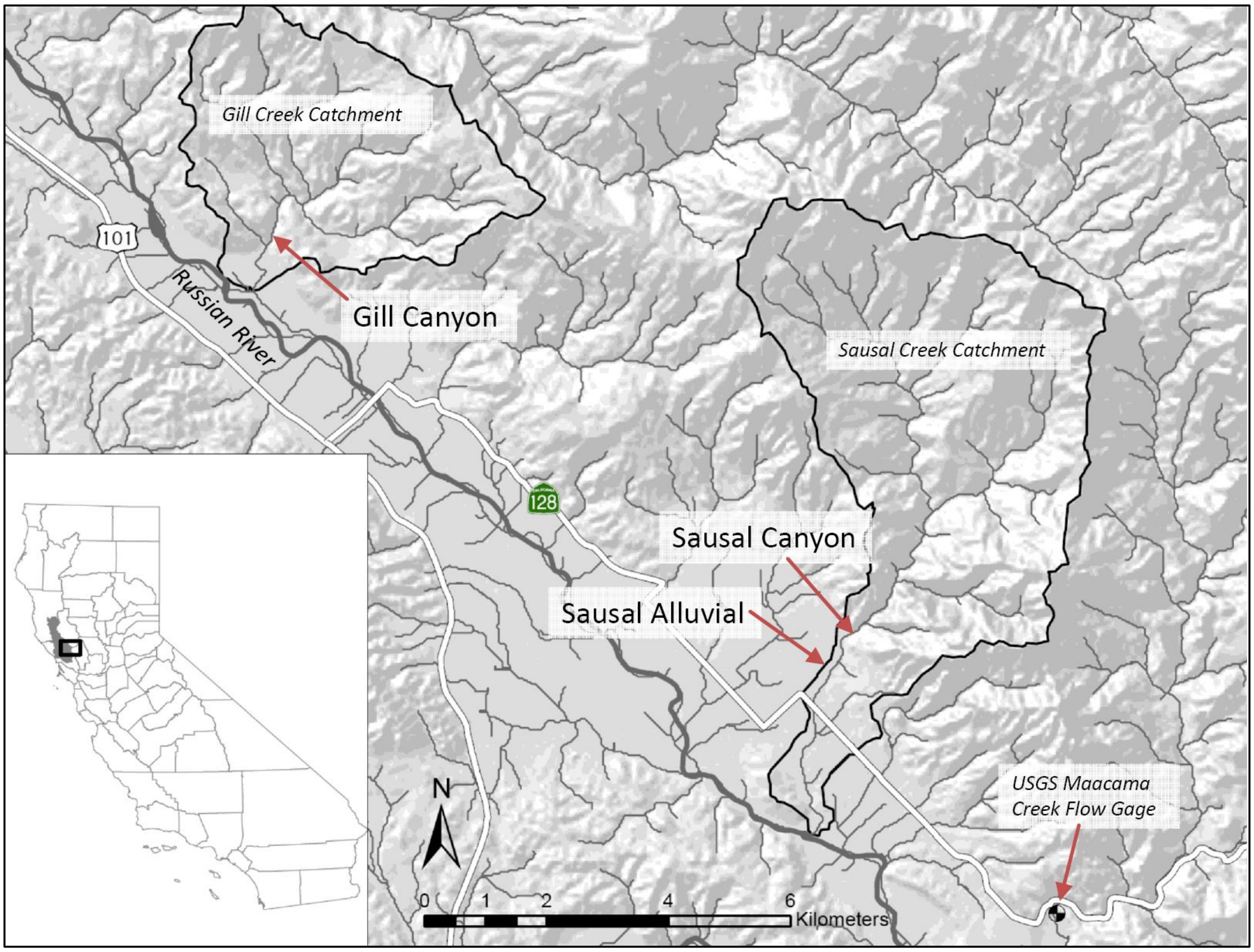


Figure 4.2 Sample cross sections of Gill Canyon, Sausal Canyon, and Sausal Alluvial reaches.

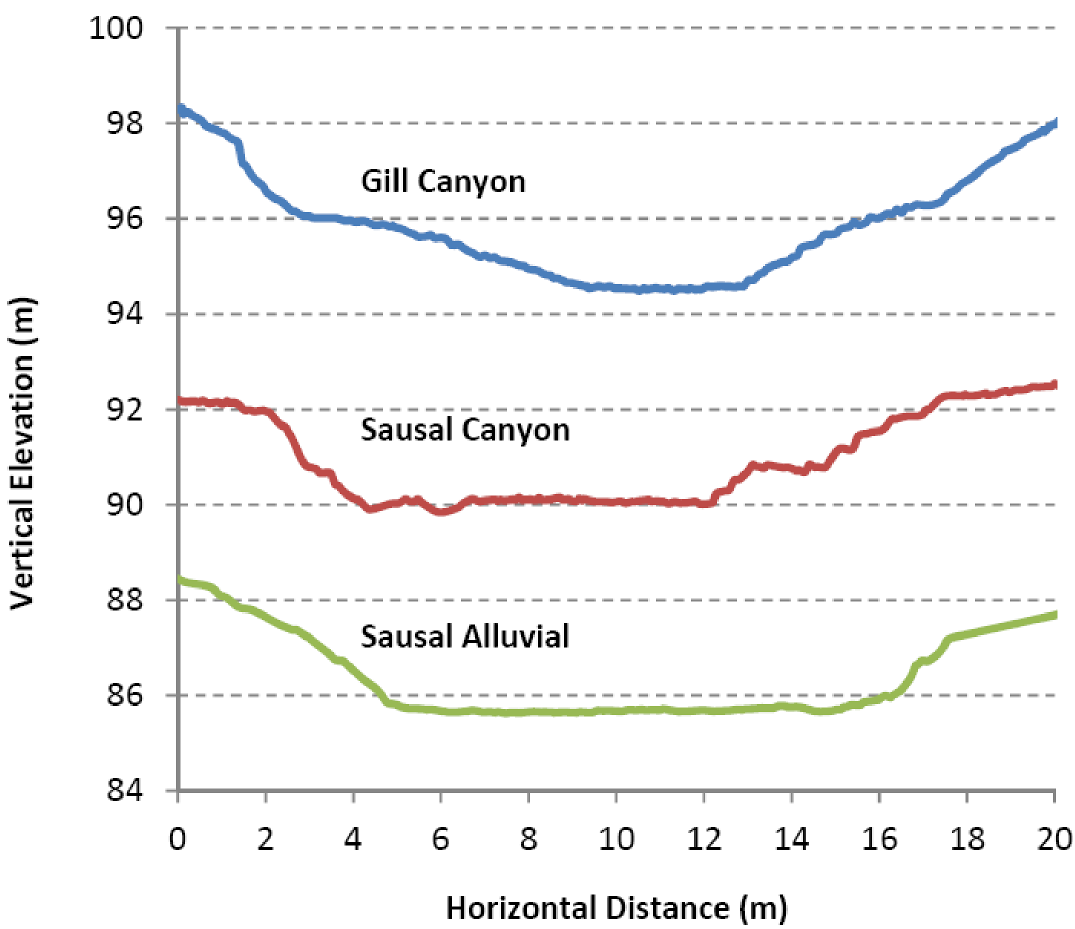


Figure 4.3 Measured versus modeled water surface elevations from calibrated MD_SWMS flow simulations.

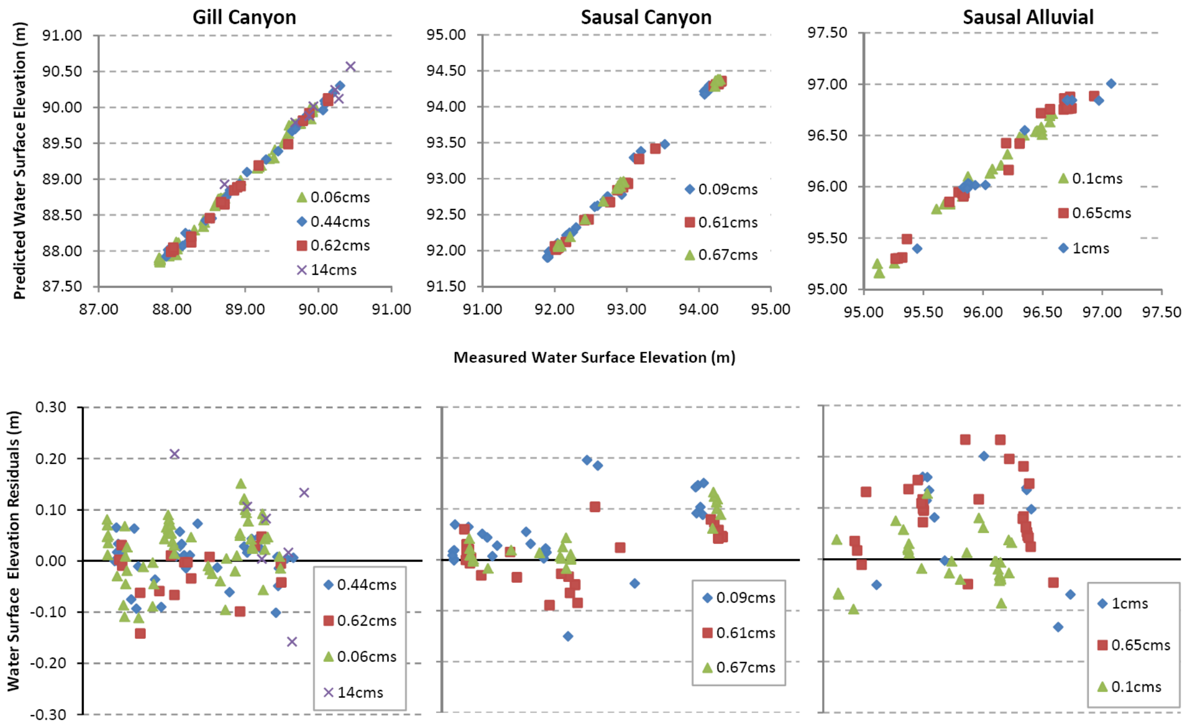


Figure 4.4 Measured versus modeled depth (A) and velocity (B) at from calibrated MD_SWMS flow simulations at 0.6 cms at all sites.

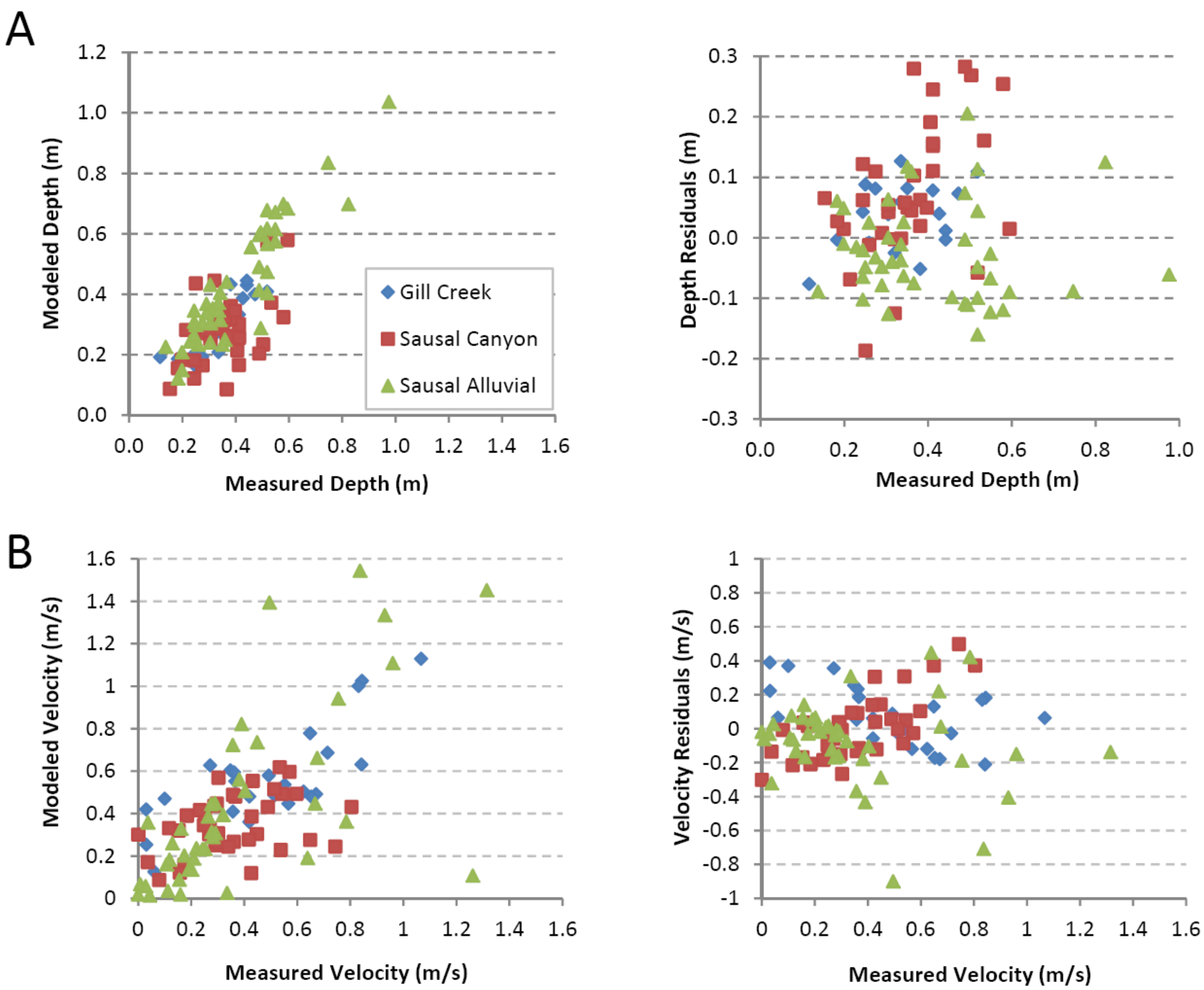


Figure 4.5 Distributions of depths and velocities scaled by wetted area within modeled study reaches.

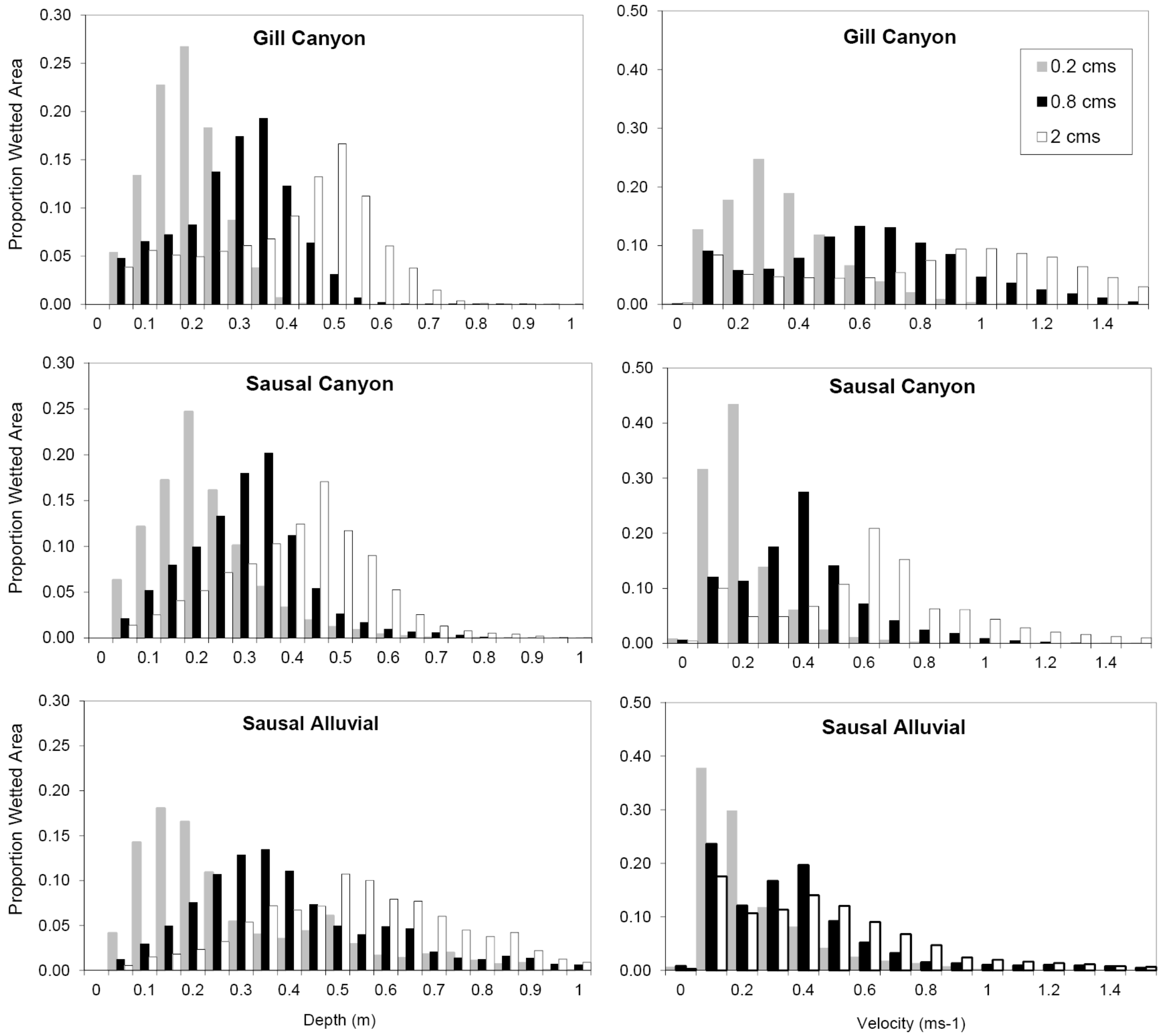


Figure 4.6 Mean depth and velocity as a function of discharge.

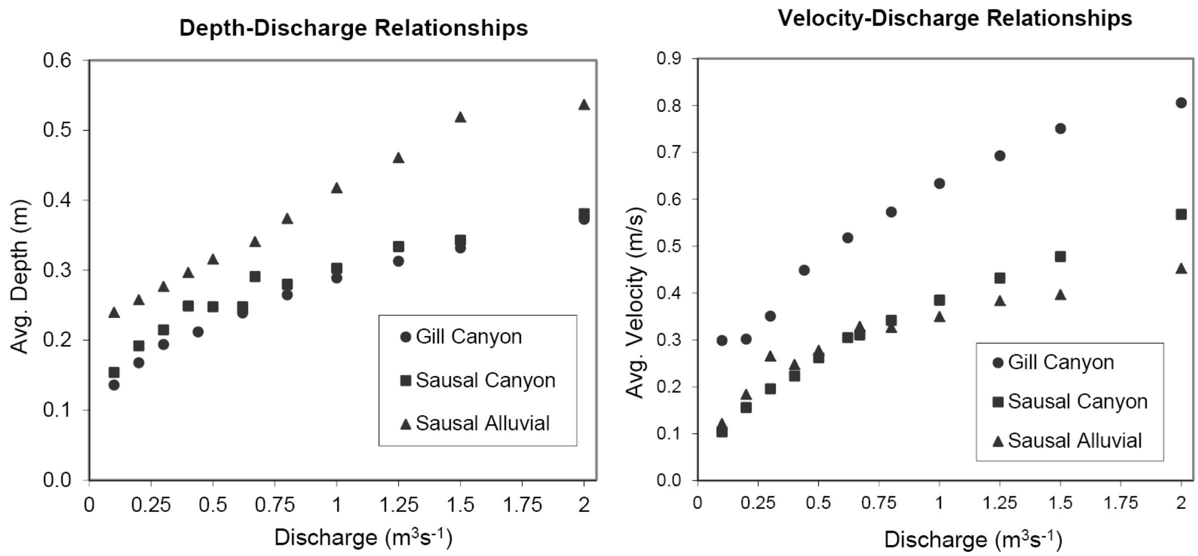


Figure 4.7 Passage flow connectivity along migration path as a function of discharge. (A) The proportion of cells equal or greater than the minimum depth criteria (0.25 m) along the migration path increases with discharge. (B) The maximum length of path segments that fall below the minimum depth criteria decreases with discharge.

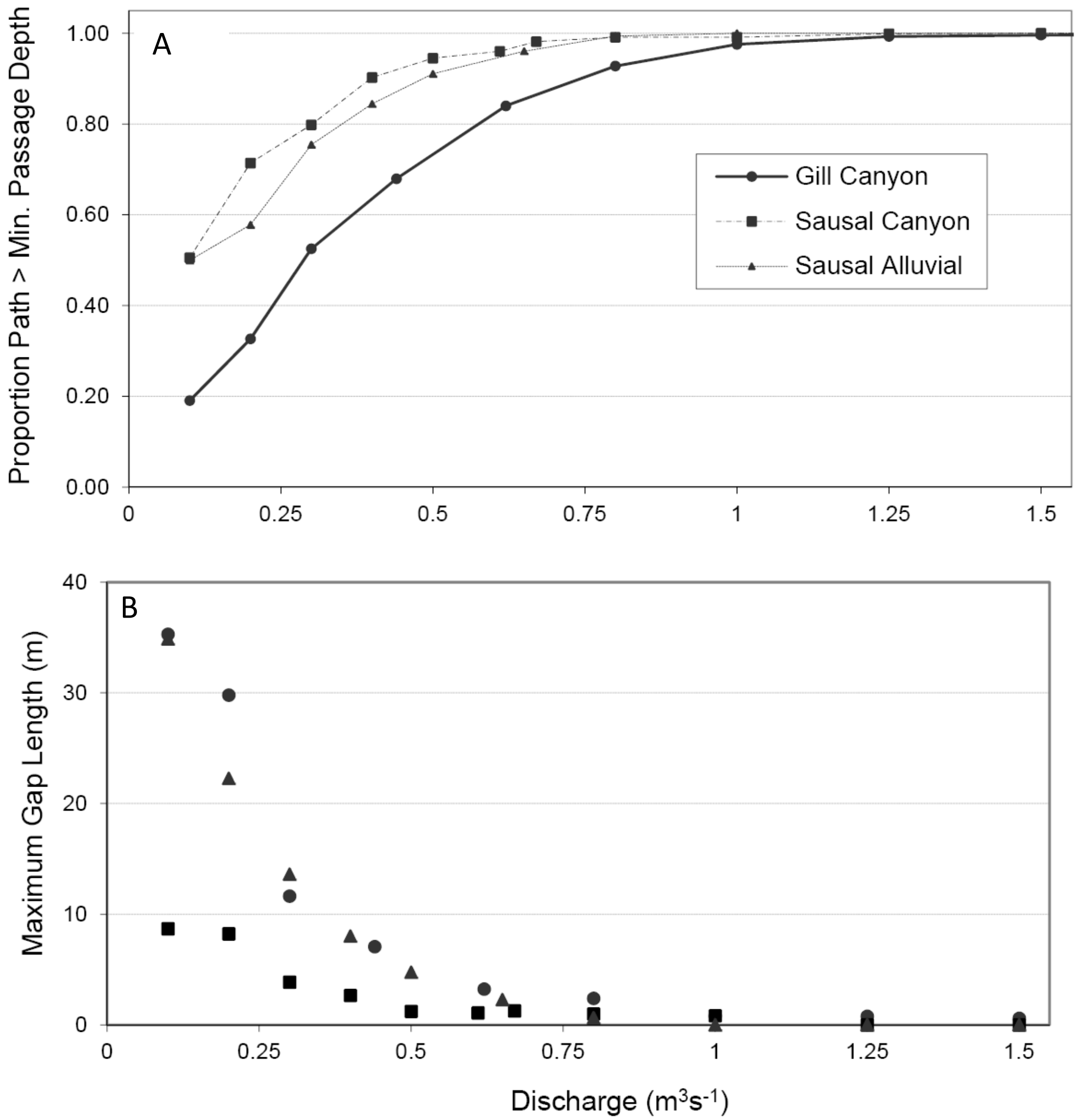


Figure 4.8 Modeled MD_SWMS depth output for in Gill Canyon reach for (A) 0.1 m^3s^{-1} (B) 0.5 m^3s^{-1}, and (C) 1 m^3s^{-1}, indicating migration path and riffle-crest transects. Cells colored in blue are at or above the 0.25 m depth criteria for fish passage.

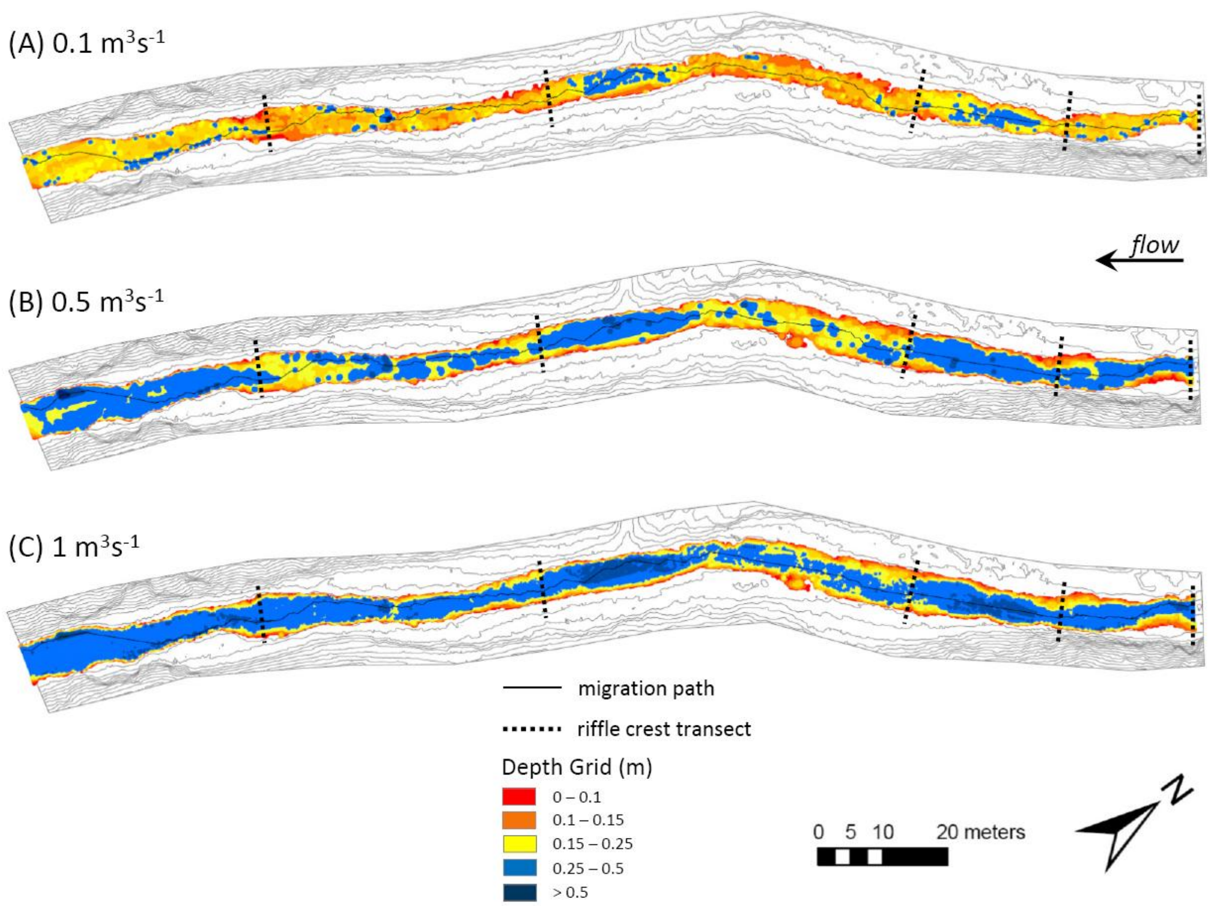


Figure 4.9 Modeled MD_SWMS depth output for in Sausal Canyon reach for (A) 0.1 m^3s^{-1} (B) 0.5 m^3s^{-1}, and (C) 1 m^3s^{-1}, indicating migration path and riffle-crest transects. Cells colored in blue are at or above the 0.25 m depth criteria for fish passage.

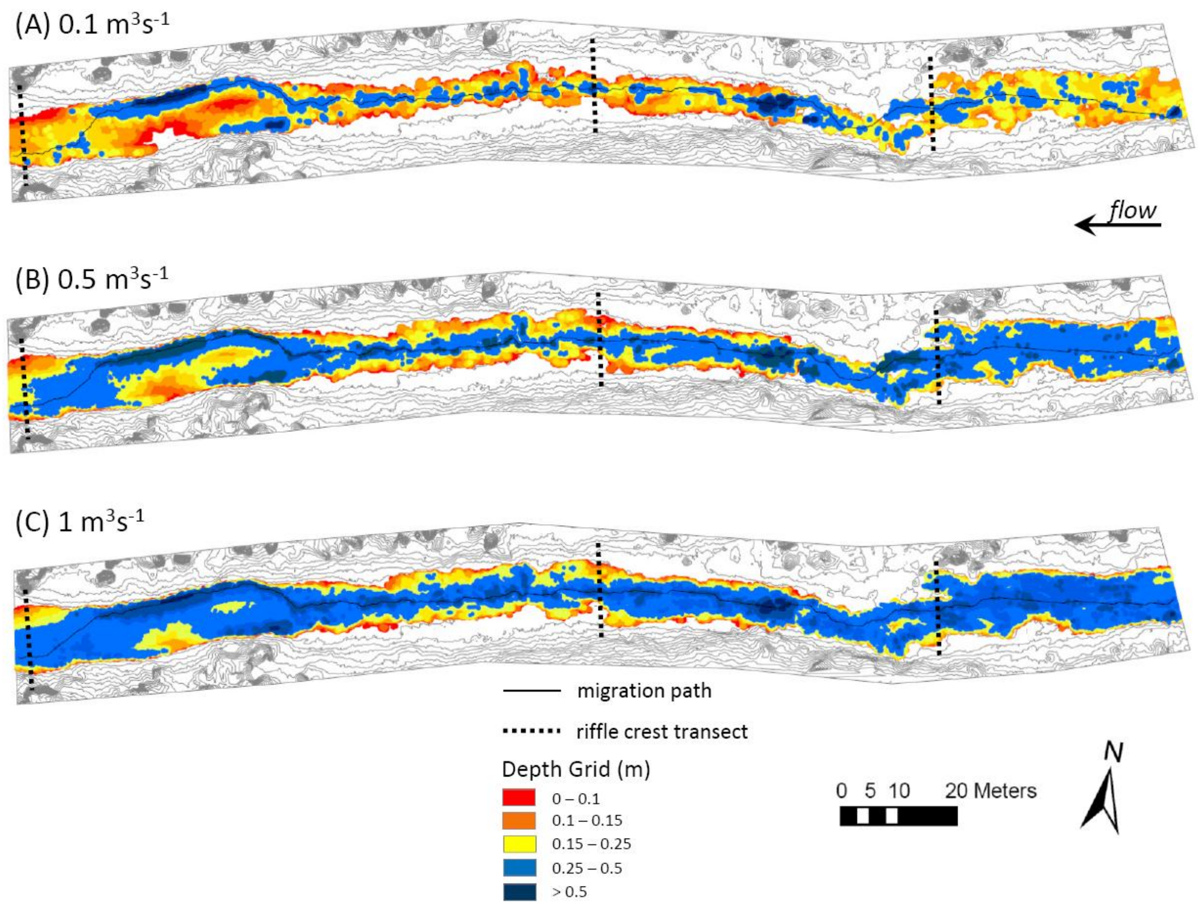


Figure 4.10 Modeled MD_SWMS depth output for in Sausal Alluvial reach for (A) 0.1 m^3s^{-1} (B) 0.5 m^3s^{-1}, and (C) 1 m^3s^{-1}, indicating migration path and riffle-crest transects. Cells colored in blue are at or above the 0.25 m depth criteria for fish passage.

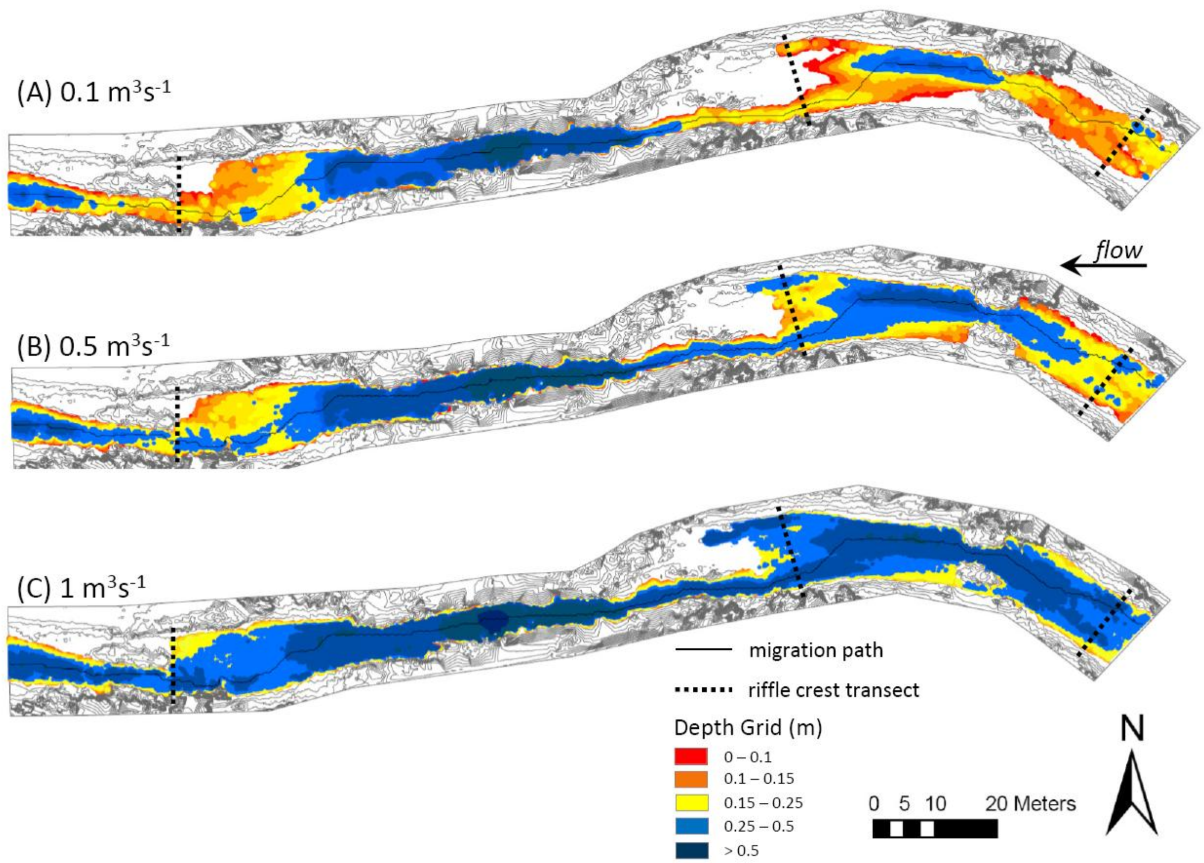
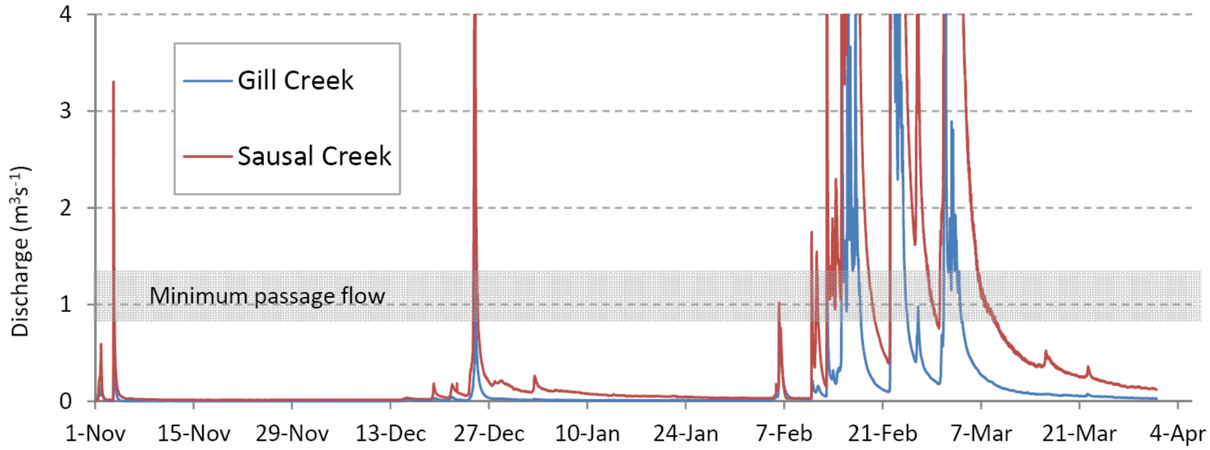


Figure 4.11 Winter hydrograph (November 1, 2008 to March 31, 2009) at Gill Creek and Sausal Creek. Shaded bar indicates the range of flow requirements for fish passage at the three study sites, based on 2-D hydraulic modeling simulations.



CHAPTER 5

CONCLUSIONS AND FUTURE RESEARCH

Conclusions and Future Research

The research presented in this dissertation investigates the relationships between hydrology, ecology, and human water demands in the management of freshwater ecosystems in Mediterranean-climate California. I analyzed the environmental factors controlling the abundance and survival of juvenile salmon populations, introduced a framework for evaluating the effects of alternative water management strategies on stream ecosystems, and developed a new approach for assessing passage flow requirements of adult salmon in natural stream channels. My methods included ecological data analysis, watershed hydrologic modeling, stream flow gaging, habitat surveys, and hydraulic modeling. The overall objectives of the research presented are to advance understanding of the linkages between hydrologic dynamics and ecological functions, to improve restoration outcomes, and support the development of sustainable water management practices. Given the public resources invested in the recovery of salmon populations and the growing human pressures on freshwater resources, future research should continue to examine the effects of water use practices on watershed hydrology and the consequences of flow regime alterations on the integrity of stream ecosystems.

Results from the data analysis in Chapter 2 indicate that summer stream flows may be a limiting factor to rearing juvenile salmon populations. However, the mechanisms by which flow influences the survival and fitness of individual salmon in the dry season are poorly understood. Stream flow largely regulates the input of invertebrate drift, a primary food sources for trout which potentially influences the fitness and survival of individual fish (Hayes et al. 2008). Thus, the relationship between flow dynamics and food supplies in the low-flow season warrants further study. Flow patterns are also tightly coupled with stream thermal regimes and water quality parameters, although the relationships are complicated by sub-surface flow dynamics that are difficult to measure. An improved understanding of the role of hyporheic flows for maintaining suitable habitat conditions for rearing salmon would be a significant contribution to our understanding of Mediterranean stream ecosystem dynamics. Finally, there is uncertainty about the extent to which juvenile salmon are able to seek and find habitat refugia during the on-set of the dry season. Identifying the environmental factors that trigger, or limit, individual fish movement would help to distinguish mortality from emigration in time series analyses and would have important implications for conservation (e.g., determining if and when to capture and relocate fish).

Mediterranean-climate watersheds are characterized by high spatial and temporal variation in stream habitat dynamics, which presents formidable challenges to making robust scientific inferences about ecological-flow relationships. Furthermore, the linkages between hydrologic and ecological dynamics are mediated by processes operating at multiple scales, from the stream reach to the watershed and geographic region (Magalhães et al. 2007). Long-term data are needed to distinguishing the effects of multiple natural and anthropogenic stressors on freshwater ecosystems (Bêche et al. 2009). In the north coast region of California, a wide variety of government agencies and non-profit groups have invested resources towards monitoring river flows, water quality, stream habitat conditions, fish populations, and aquatic macroinvertebrate communities. Yet monitoring efforts are sporadic, limited in spatial extent, and poorly coordinated. The integration of environmental monitoring efforts and establishment of consistent protocols would vastly increase the utility of the data and create new opportunities for detecting patterns and revealing potential causal mechanisms between ecological and physical environmental conditions.

Although there is evidence that privately operated, small-scale water diversions can reduce stream flows and potentially impair aquatic habitat (Deitch et al. 2009), our understanding the effects of water diversions on the hydrology and ecology of streams is limited. Additional research on water diversion operations, their individual and cumulative effects on flow regimes, and the consequences of flow alterations to habitat suitability is needed. Assessing human alterations to the hydrology of Mediterranean watersheds is complicated by the geologic heterogeneity and climate variability that affect the surface flows. Because the effects of withdrawals are not easily separated from natural flow dynamics in the low-flow period, quantifying the influence of human diversions on streams that naturally reach intermittency in the dry season is challenging. Manipulative studies, in which the timing and rates of water diversions could be controlled, are probably necessary for detecting the impacts of water diversions, particularly during the dry season. Ultimately, such information could be integrated with watershed management models, as described in Chapter 3, to more accurately represent the potential effects of water users distributed across the stream network.

Advances in hydraulic modeling offer new opportunities for investigating the ecological effects of flow alterations at fine spatial scales. The two-dimensional hydraulic modeling approach presented in Chapter 4 accurately simulates the spatial distribution of depths, velocities and other hydraulic parameters in stream reaches at high resolution that would be infeasible to reproduce through synoptic measurements. Linking calibrated hydraulic model predictions with detailed observations of fish movement and occupancy would make it possible to examine the influence of flow-mediated habitat variables on biological responses. Therefore, high resolution hydraulic modeling offers a promising means of advancing research and understanding at the biological-physical interface.

The study and management of river ecosystems remains highly fragmented across disciplines, including hydrology, ecology, geomorphology, flood risk management, and civil engineering (Vaughan et al. 2009). Meanwhile, widespread indicators of ecological degradation, biodiversity loss, and impairment of valuable ecosystem services suggest that prevailing approaches for managing freshwater resources are woefully inadequate. Projected climate change and population growth in Mediterranean-climate and other semi-arid regions will unquestionably increase pressures on water resources and intensify impacts to freshwater ecosystems already in severe decline. Thus, interdisciplinary research efforts, such as presented in this dissertation, are urgently needed to advance scientific understanding and guide the management of freshwater ecosystems.

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