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Authors
Moniz, Peter J
Pasternack, Gregory B

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Chinook salmon rearing habitat-discharge relationships change as a result of morphodynamic processes

Peter J. Moniz*, Gregory B. Pasternack

Department of Land, Air, and Water Resources, University of California, Davis, Davis, California, USA

*Corresponding author email: pjmoniz@ucdavis.edu

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Abstract

Relationships between fluvial aquatic habitat availability and discharge are often assumed to remain static when used with hydrologic datasets to analyze changes in habitat availability over time. Despite this assumption, studies have observed significant changes in aquatic habitat availability before and after restoration projects, dam removals, and extreme flood events. However, research is lacking on how aquatic habitat changes as a result of morphodynamic processes during more commonly occurring hydrologic conditions. This study compared Chinook salmon (*Oncorhynchus tshawytscha*) rearing habitat availability at 19 discharges before and after a relatively mild 8-year hydrologic period punctuated with modest floods on the lower Yuba River in California, USA. During this time, the total area of rearing habitat remained relatively consistent at discharges <2x bankfull but decreased by up to 25% at discharges >2x bankfull. Significant decreases in rearing habitat area appeared to be the result of widespread erosion on floodplains, terraces, and lateral bars, even after only modest floods. As a result, spatially delineated areas of lost habitat tended to increase in water depth and velocity at baseflow, bankfull, and floodplain-filling discharges, while areas of gained habitat decreased in depth and velocity. Although these specific results may not apply to all rivers around the world, the finding that habitat-discharge relationships change as a result of morphodynamic processes likely does transfer globally and should be considered when making long-term regulatory and management decisions, such as instream flow requirements and habitat restoration plans.

KEYWORDS

aquatic habitat, habitat change, habitat suitability model, rearing habitat, salmonid habitat

INTRODUCTION

Habitat suitability models have become a common tool used by resource managers to analyze how changes in discharge, substrate, and/or channel topography relate to the abundance and quality of aquatic habitat in river systems (Ahmadi-Nedushan et al., 2006; Dunbar, Alfredsen, & Harby, 2012). These models typically assign relative indices of habitat quality (i.e., suitability values) to spatially explicit maps of physical habitat values (e.g., water depth, water velocity, substrate size, and cover type). Suitability values range from 0 (least suitable) to 1 (most suitable) (Bovee, 1986) and are linked to physical habitat values using a variety of methods, including expert judgment, correlative statistics, and bioenergetics (Ahmadi-Nedushan et al., 2006; Dunbar et al., 2012; Naman, Rosenfeld, Neuswanger, Enders, & Eaton, 2019; Rosenfeld, Beecher, & Ptolemy, 2016). Once developed, habitat suitability models are often used to calculate a total habitat area (or "habitat availability") at a range of discharges and/or with multiple restoration design alternatives and are used to inform regulatory and management decisions.

Habitat-discharge relationships developed from habitat suitability models are often applied to actual or theoretical hydrologic time-series data in an attempt to analyze changes in habitat availability over time (Benjankar et al., 2018; Boavida, Caetano, & Pinheiro, 2020; Stamou et al., 2018). However, this method assumes that although the physical habitat conditions within a river may change over time, the relationship between habitat availability and discharge remains static throughout the analyzed time period. Despite such commonly held assumptions, multiple studies have observed changes in habitat-discharge relationships before and after habitat restoration projects (Gard, 2006; Harrison, Legleiter, Wydzga, & Dunne, 2011; Hauer, Unfer, Holzmann, Schmutz, & Habersack, 2013; Wheaton et al., 2010), dam removals (Im, Kang, Kim, & Choi, 2011; Mouton, Schneider, Depestele, Goethals, & De Pauw, 2007; Tomsic, Granata, Murphy, & Livchak, 2007), and extreme flood events (Gard, 2014; Harrison, Pike, & Boughton, 2017; Mandlburger, Hauer, Wieser, & Pfeifer, 2015; Tamminga & Eaton, 2018). However, research is lacking on the stability of aquatic habitat-discharge relationships without major anthropogenic or hydrologic disturbances, particularly at scales larger than a specific project site. Changes in habitat availability induced by fluvial morphodynamics stand out as a particularly important, yet uninvestigated topic in river research and applications. Furthermore, studies related to this topic often lack in-depth analysis of spatial patterns of habitat change and how physical habitat conditions (e.g., depth, velocity, cover) change at specific locations within the river over time.

The purpose of this study was to evaluate the stability of Chinook salmon (*Oncorhynchus tshawytscha*) fry (i.e., visually observed length < 50 mm) habitat availability in a regulated yet dynamic gravel-cobble river during a relatively mild hydrologic period with modest flooding. The novelty and importance of this study lies in introducing new methods for evaluating changes in habitat availability and linking those changes to morphodynamically induced changes in specific physical habitat conditions. Note that this study does not investigate morphodynamic processes explicitly, but the response of habitat availability to such processes. Specific questions addressed in this study include:

1. Did key metrics in the habitat-discharge relationship, such as the minimum and maximum habitat availability change in response to morphodynamic changes?
2. Were morphodynamically induced changes in habitat availability greater at in-channel or overbank discharges?
3. How did areas of lost, sustained, and gained habitat vary in patch size and longitudinal distribution throughout the river at baseflow, bankfull, and floodplain-filling discharges?
4. Could changes in Chinook salmon fry rearing habitat availability be explained by changes in channel alignment, hydraulic conditions, and/or cover?

STUDY SITE AND HYDROGEOMORPHIC SETTING

The Yuba River is a tributary of the Sacramento River in northern California that drains 3,480 km^2 of the western slopes of the Sierra Nevada (Figure 1). The lower Yuba River (LYR), defined as the 37-km segment of the river between Englebright Dam and the Feather River confluence, is a regulated gravel-cobble bed river with a low sinuosity, high width-to-depth ratio, and slight to no entrenchment (Wyrick & Pasternack, 2014). It includes critical habitat for Central Valley spring-run Chinook salmon, currently listed as threatened under the United States Endangered Species Act (National Marine Fisheries Service, 2014; US Fish and Wildlife Service, 2010). As one of the most fluvially dynamic regulated rivers in the coterminous United States, the LYR is an ideal testbed for studying morphodynamically induced habitat change.

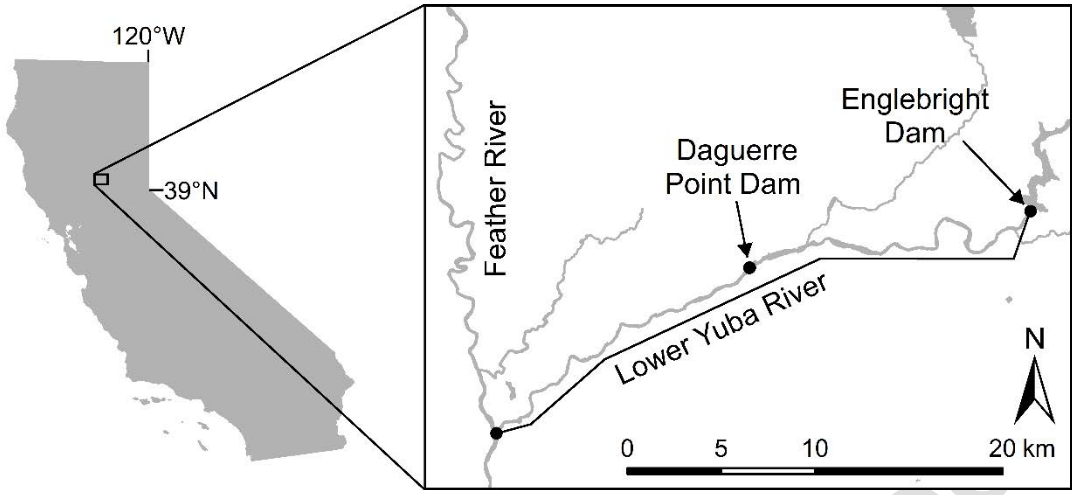

FIGURE 1 Map of the study location in the lower Yuba River

Whereas the common expectation for a regulated river is a relatively static geomorphic state with reduced channel-floodplain connectivity, the LYR is very different. From 1999–2014, it experienced millions of cubic meters of gross erosion and deposition (Carley et al., 2012; Weber & Pasternack, 2017), differentiated among landforms and facilitated by 19 different morphodynamic processes functioning valley wide (Wyrick & Pasternack, 2015). Further, the bankfull channel experienced net fill during this time, while the overbank region experienced net erosion, enhancing channel-floodplain connectivity (Weber & Pasternack, 2017). This dynamism is driven by a relatively undisturbed flood regime acting on a massive, anthropogenically derived sediment supply stored below Englebright Dam (Gilbert, 1917; James, 2005).

Discharge into LYR primarily comes from three tributaries: the North, Middle, and South Yuba Rivers. Although the North Yuba River has a large reservoir that heavily regulates its outflow year-round, the absence of large reservoirs on the Middle and South Yuba Rivers allows for hydrologically dynamic conditions in the LYR. Relatively frequent floods occur when Englebright Dam is overtopped during winter storms and spring snowmelt and are capable of driving substantial morphodynamic change (Sawyer, Pasternack, Moir, & Fulton, 2010). Daguerre Point Dam (DPD) is an 8-m-high run-of-the-river, sediment-retention dam that also aids flow diversion for irrigation. It is located near the middle of the LYR, 17.8 river kilometers upstream from the Feather River. Storage behind DPD is filled with sediment, allowing bedload to pass downstream during floods.

The 8-year hydrologic period covered in this study spanned the 2007 through 2014 water years and was considered relatively dry in California. Seven of the 8 years were classified as extremely critically dry, dry, or below normal according to the Yuba River Index, which is calculated based on the current and previous year's unimpaired flows (Yuba County Water Agency, 2012). During this time, the LYR experienced only four modest ~3–5 year flood events of short duration and a maximum instantaneous discharge of 9x bankfull (Figure 2). The total duration of flood events quickly dropped with flood magnitude, with 171.3 days of discharges above bankfull, 28.7 days above 2x bankfull, and 4.4 days above floodplain-filling flow. Despite the relatively mild hydrologic conditions, Weber and Pasternack (2017) observed significant topographic change on the LYR during this time, raising the question as to whether and how such changes affected habitat availability.

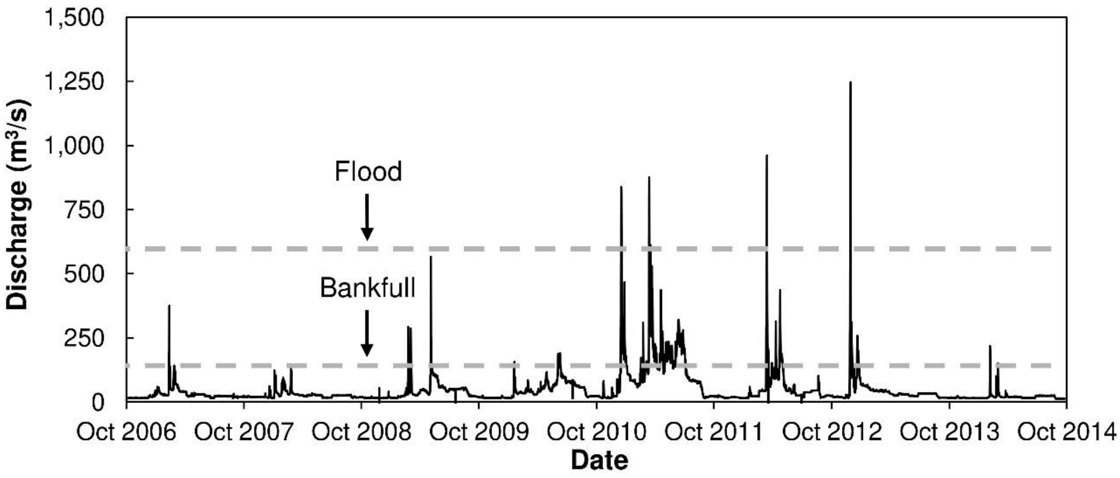

FIGURE 2 Hydrograph of instantaneous discharge in the lower Yuba River from 2006 to 2014 at the Marysville USGS gage (11421000). Bankfull and flood (i.e., floodplain-filling) discharges were defined by Wyrick and Pasternack (2012) as 141.6 and 597.5 m^3/s, respectively

PRECUSORY MODELS

The habitat suitability models used in this study were developed using results of several previously published models of the LYR. A brief summary of the development and evaluation of these precursory models is provided below, with relevant details and references presented as a summary table in the Supporting Information document. A detailed description of the novel methods used in this study to evaluate and explain changes in Chinook salmon fry rearing habitat availability in the LYR is then provided in the following section. Spatial analyses used in this study were performed using ArcGIS v. 10.6 (ESRI, 2018).

To characterize the topo-bathometric conditions of the LYR at the beginning and the end of the study period, two 0.91-m-resolution digital elevation models (DEMs) were developed using a combination of ground-based surveying, boat-based bathymetry, and airborne LiDAR collected in 2006/2008 and 2014. The 2006/2008 model was a combination of 2 years of topo-bathymetric data, with the uppermost 20% of the LYR's length mapped in 2006 and the other 80% mapped in 2008. Survey and interpolation errors estimated for both DEMs for bare ground, water, and vegetated ground were 0.039, 0.074, and 0.30 m, respectively (Weber & Pasternack, 2017). Details of 2006/2008 and 2014 DEM development and uncertainty evaluation can be found in Carley et al. (2012) and Weber and Pasternack (2017), respectively.

From these DEMs, two-dimensional (2D) hydrodynamic models were produced for the study site using SHR-2D for 2006/2008 hydraulic conditions and TUFLOW GPU for 2014 conditions. A model comparison study of 2014 river conditions found minimal difference between these solvers for this river (Pasternack & Hopkins, 2017). The result of these models was two sets of 0.91-m-resolution depth and velocity rasters for the LYR. Extensive hydrodynamic validation found that the performance of both models far exceeded peer-reviewed journal norms (Pasternack, 2011). For example, the coefficient of determination (R^2) between predicted and observed values for depth, velocity magnitude, and velocity direction were >0.65 for both models. Details of 2006/2008 and 2014 hydrodynamic model development and validation can be found in Barker et al. (2018) and Hopkins and Pasternack (2018), respectively.

Cover features were digitized into 0.91-m rasters throughout the study site. Vegetation presence/absence was mapped using airborne LiDAR collected in 2008 (Abu-Aly et al., 2014; Burman & Pasternack, 2017) and 2014 (Weber & Pasternack, 2016; Weber & Pasternack, 2017). Permeant human-built detritus (e.g., rip-rap, cement blocks) were mapped using a combination of 0.3-m-resolution aerial imagery from 2008 and boat-based field reconnaissance in 2012 (Vaughan & Pasternack, 2014). Other permanent cover features (e.g., bedrock outcrops, weirs, bridge piers) were mapped using 0.3-1-m resolution aerial imagery from 2008 and 2012. Detailed field surveys were conducted in 2010 and 2011 confirming widespread cover of large cobble substrate throughout the LYR (Jackson, Pasternack, & Wyrick, 2013).

Habitat suitability criteria (HSC) functions for depth, velocity, and cover type were developed and bioverified for Chinook salmon fry by Moniz, Pasternack, Massa, Stearman, and Bratovich (2020). HSC functions were applied to 2014 depth, velocity, and cover rasters, resulting in a set of univariate depth, velocity, and cover habitat suitability rasters at a range of discharges between 14.16 and 42.48 m^3/s. These rasters were then combined cell-by-cell using the geometric mean function, resulting in a combined habitat suitability raster of the entire LYR for each modeled discharge. Moniz et al. (2020) used bootstrapped electivity indices to classify ranges of suitability values into preferred, avoided, and randomly selected microhabitat. Mathematically, preferred microhabitat was defined as a range of suitability values in which fish were observed at a disproportionally higher percentage than the area of river available having that same range of suitability values. Statistical bootstrapping was used to calculate a 95% confidence interval for each of these ranges to more accurately differentiate preferred and avoided microhabitat from randomly selected microhabitat. Details of the development and bioverification of HSC functions and 2014 habitat suitability models can be found in Moniz et al. (2020).

METHODS

Delineating preferred microhabitat

To evaluate and explain changes in Chinook salmon fry rearing habitat availability in the LYR, this study applied the HSC functions developed and bioverified by Moniz et al. (2020) to a subset of the 2006/2008 depth, velocity, and cover rasters summarized above. Using the preference threshold observed by Moniz et al. (2020) for Chinook salmon fry in the LYR, cells in the 2006/2008 and 2014 habitat suitability rasters with suitability values ≥0.5 were classified as preferred microhabitat and converted into polygon features to be further analyzed in this study.

Minor modifications had to be made to the 2014 preferred microhabitat polygons to make them comparable to the 2006/2008 polygons. Because the 2014 LiDAR coverage extended beyond the 2006/2008 coverage, 2014 models included several features (e.g., tributaries and off-channel ponds) that may have been present in 2006/2008 but were not included in the 2006/2008 models. To make the two datasets exactly comparable, 2014 microhabitat polygons were clipped by the 2006/2008 model boundary. Additionally, because the upstream canyon section of the river known as Englebright Dam Reach was not modeled for discharges between 8.50 and 16.9 m^3/s in 2006/2008, the reach was clipped from 2014 maps at these discharges.

Preferred microhabitat-discharge relationship

The total areas of preferred microhabitat were calculated for both years and plotted at 19 discharges, chosen to span a wide range of rearing conditions from 0.06 to 8.5x bankfull. Although more than 19 discharges were originally modeled for the 2006/2008 and 2014 datasets, only 18 matched between them. Those 18 matching discharges and one additional discharge representing the maximum available habitat for either dataset were selected and compared in this study. Percentage changes in overall minimum and maximum habitat availability were computed, and changes in in-channel and overbank habitat availability were compared between years. It should be noted that the same depth, velocity, and cover HSC functions were used for all 19 discharges. This assumes that Chinook salmon fry will change locations within the LYR to maintain suitable depths and velocities as discharge changes, which has been observed for other juvenile Pacific salmon species (Beecher, Carleton, & Johnson, 1995; McMahon & Hartman, 1989; Shirvell, 1990; Shirvell, 1994).

Lost, sustained, and gained preferred microhabitat

This study introduces the scientific concept and methods for computing and analyzing lost, sustained, and gained preferred microhabitat using geospatial analysis of 2D habitat suitability model results. Polygons of 2006/2008 and 2014 preferred microhabitat were clipped and intersected to create areas of lost, sustained, and gained microhabitat. Areas of preferred microhabitat available in 2006/2008 but not in 2014 were considered “lost,” while areas of preferred microhabitat not available in 2006/2008 but available in 2014 were considered “gained.” Areas of preferred microhabitat that overlapped between years were considered “sustained” microhabitat.

The total area, size, and spatial distribution of spatially explicit patches of preferred microhabitat that were lost, sustained, and gained between 2006/2008 and 2014 were evaluated at baseflow, bankfull, and floodplain-filling discharges, as defined by Wyrick and Pasternack (2012). A representative baseflow discharge above and below DPD was defined to be 24.92 and 15.01 m^3/s, respectively, with the difference accounting for irrigation withdrawals. Bankfull and floodplain-filling (hereafter referred to as “flood”) discharges were defined to be 141.6 and 597.5 m^3/s, respectively.

These discharge-dependent areas of lost, sustained, and gained preferred microhabitat were compared for the entire LYR in three ways. First, the total area of lost, sustained, and gained preferred microhabitat was compared for the entire LYR. Second, individual polygons of lost, sustained, and gained microhabitat were grouped by size, and the total areas of each polygon type were compared by group. Third, longitudinal profiles of the cumulative areas of lost, sustained, and gained microhabitat were compared to better understand their distribution throughout the river. This was done by computing the areas of lost, sustained, and gained microhabitat within discrete cross-sectional rectangles stationed every 6.1 m (20 ft) along the river valley’s centerline (methods in Pasternack & Wyrick (2017)).

Changes in physical habitat conditions associated with lost, sustained, and gained microhabitat

To determine whether changes in Chinook salmon fry rearing habitat availability could be explained by changes in specific physical habitat conditions, more detailed analyses of lost, sustained, and gained microhabitat were performed at baseflow, bankfull, and flood discharges. Changes in channel alignment were examined by computing percent differences in wetted area between 2006/2008 and 2014, the percentages of 2006/2008 microhabitat lost by becoming dry land, and the percentages of 2014 microhabitat gained by dry land becoming inundated.

To determine whether changes in individual physical variables could explain changes in habitat availability, numerical differences in depth, velocity, and cover were computed between 2006/2008 and 2014 within areas of lost, sustained, and gained microhabitat. Specifically, 2006/2008 depth and velocity rasters were subtracted cell-by-cell from their respective 2014 rasters. Rasters of these differences were then clipped by the lost, sustained, and gained polygons associated with each discharge and grouped by every 0.2 m and m/s, respectively. The total area of lost, sustained, and gained microhabitat within each grouped difference was then computed. Given that permanent instream cover features (e.g., bedrock outcrops, rip-rap, bridge piers, and weirs) could not change between years, cell-by-cell differences in cover conditions could be associated with either no change, a loss, or a gain in vegetative cover, which could change over time. To better understand these changes, percentages of areas where there was a loss, gain, or no change in vegetative cover were computed for lost, sustained, and gained Chinook salmon fry rearing microhabitat polygons.

RESULTS

Preferred microhabitat-discharge relationship

Preferred microhabitat-discharge relationships for Chinook salmon fry rearing showed the same general pattern between years but had different minimum and maximum areas of preferred microhabitat at different discharges (Figure 3). Results reflect a noticeable increase in in-channel preferred microhabitat habitat area and a decrease in overbank habitat area. From 2006/2008 to 2014, the minimum area of preferred microhabitat increased by 18% (from 27.7 ha at 36.8

m^3/s to 32.7 ha at 56.6 m^3/s). Meanwhile, the maximum area of preferred microhabitat decreased by 15% (from 72.2 ha at 424.8 m^3/s to 61.2 ha at 453.0 m^3/s). Areas of preferred microhabitat were slightly higher in 2014 at discharges below 56.3 m^3/s and remained relatively similar at discharges between 56.3 and 283.2 m^3/s, or 2x bankfull. However, a significant divergence in preferred microhabitat occurred at discharges above 283.2 m^3/s, with more habitat available in 2006/2008 than 2014.

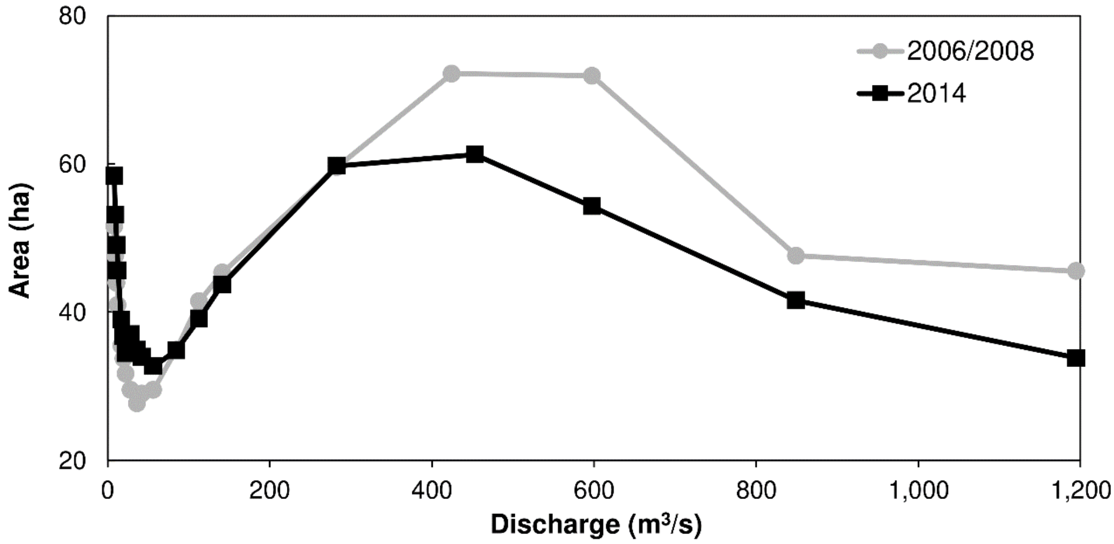

FIGURE 3 Preferred Chinook salmon fry rearing microhabitat area as a function of discharge during 2006/2008 and 2014

Lost, sustained, and gained preferred microhabitat

There was 5% more Chinook salmon fry rearing preferred microhabitat gained from 2006/2008 to 2014 than lost at baseflow discharge (Figure 4). However, during the same time period, the total area of lost habitat was 27% and 83% higher than the area of gained habitat at bankfull and flood discharges, respectively. The total area of lost and sustained microhabitat increased with discharge, while the total area of gained microhabitat remained relatively consistent.

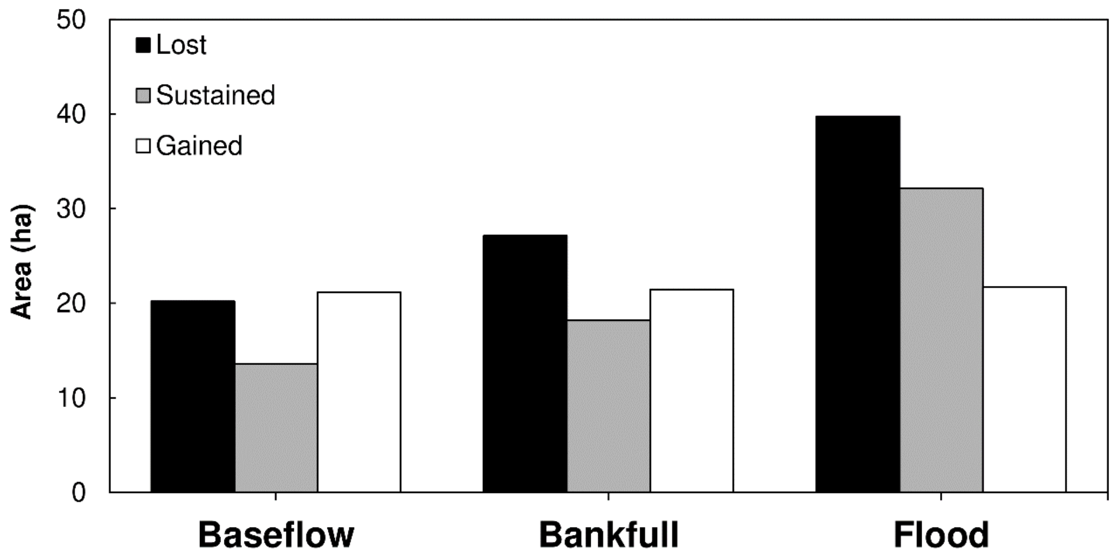

FIGURE 4 Changes in preferred Chinook salmon fry rearing microhabitat at the segment scale between 2006/2008 and 2014

Total areas of lost, sustained, and gained preferred microhabitat were primarily made up of habitat patches between 10^1–10^2, 10^2–10^3, and 10^3–10^4 m^2 at all three discharges (Figure 5). At baseflow discharge, the total areas of lost and gained microhabitat patches between 10^1–10^2 and 10^3–10^4 m^2 were relatively similar. However, the total area of gained microhabitat patches between 10^2 and 10^3 m^2 was 14% higher than the total area of lost patches. At bankfull discharge, the total areas of lost microhabitat patches between 10^1–10^2 and 10^2–10^3 m^2 were 29% and 47% higher than the total areas of gained patches, respectively, while the total area of lost and gained patches between 10^3 and 10^4 m^2 were relatively similar. At flood discharge, the total areas of lost microhabitat patches between 10^1–10^2, 10^2–10^3, and 10^3–10^4 m^2 were 29%, 151%, and 96% higher than the total areas of gained microhabitat patches, respectively. At baseflow and bankfull discharges, the total area of sustained microhabitat was highest for patches between 10^2 and 10^3 m^2, but highest for patches between 10^3 and 10^4 m^2 at the flood discharge. The 4.6 ha of sustained microhabitat patches between 10^4 and 10^5 m^2 were composed of only three patches at the flood discharge.

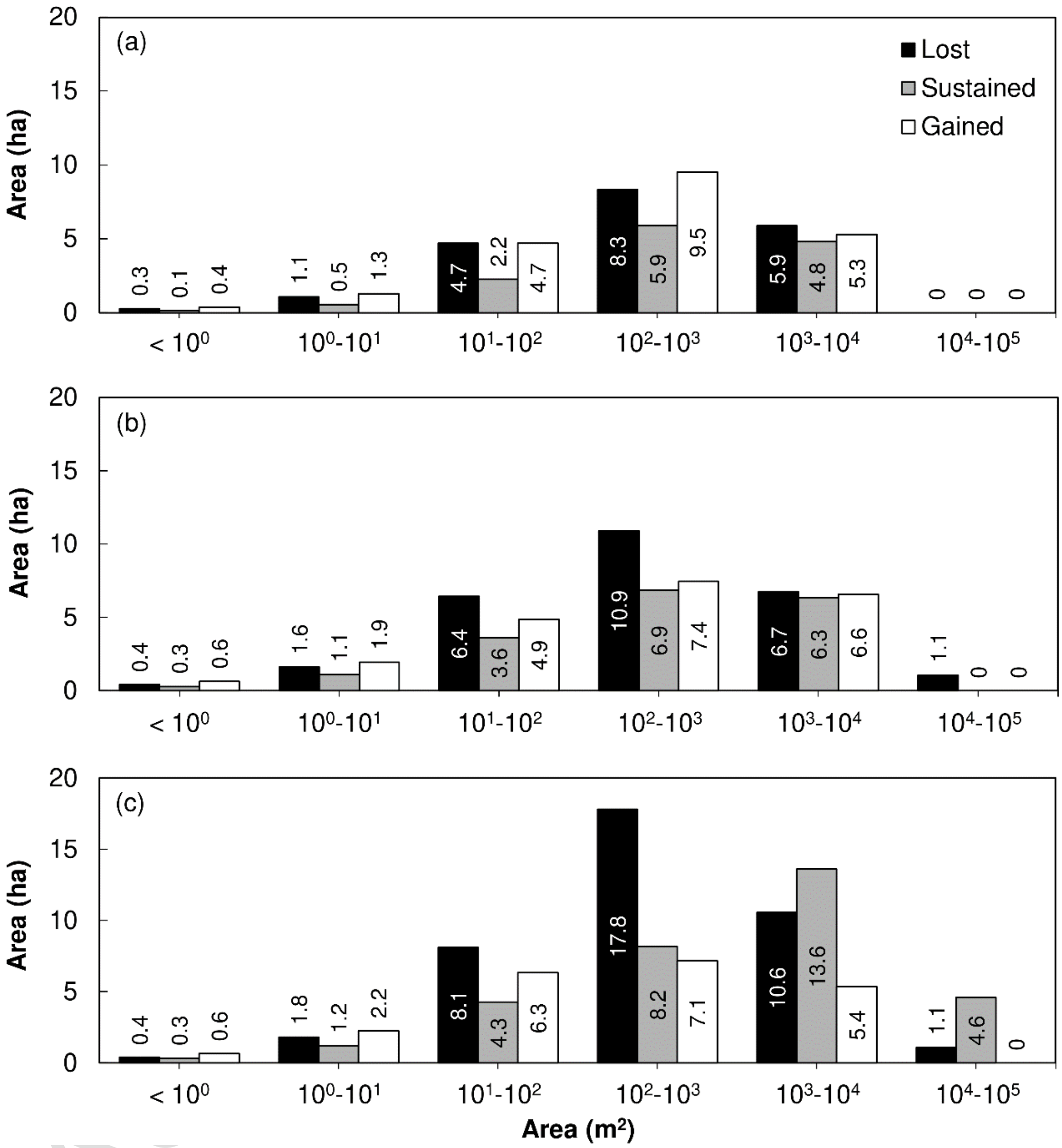

FIGURE 5 Size distributions of preferred Chinook salmon fry rearing microhabitat lost, sustained, and gained between 2006/2008 and 2014 at (a) baseflow, (b) bankfull, and (c) flood discharges

Longitudinal cumulative areas of lost, sustained, and gained microhabitat varied across the three discharges (Figure 6). At baseflow discharge, the cumulative area of lost, sustained, and gained microhabitat all steadily increased upstream until around river-km 14, where the cumulative area of sustained microhabitat flattened out relative to lost and gained microhabitat. At bankfull discharge, the cumulative area of lost, sustained, and gained microhabitat remained relatively flat between river-km 0 and 7 and then increased at approximately the same rate between river-km 7 and 14. Upstream of this point, the cumulative area of lost microhabitat increased at a faster rate than that of gained and sustained microhabitat, with the cumulative area of gained microhabitat slightly outpacing sustained microhabitat upstream of river-km 22. At the flood discharge, there were more distinct “steps” in cumulative areas compare to the other two discharges. These steps in cumulative area indicate more concentrated areas of lost, sustained, and gained microhabitat throughout the river segment, and are most noticeable downstream of river-km 22. Sustained microhabitat steps at river-km 10, 12, and 19 were associated with the three patches that were between 10^4 and 10^5 m^2. Besides the one step at river-km 7, the cumulative area of lost microhabitat at the flood discharge increased relatively smoothly longitudinally up the river, indicating a relatively constant loss of preferred microhabitat throughout the river segment. The rate of increasing cumulative area of lost microhabitat was highest between river-km 14 and 22 at bankfull and flood discharges.

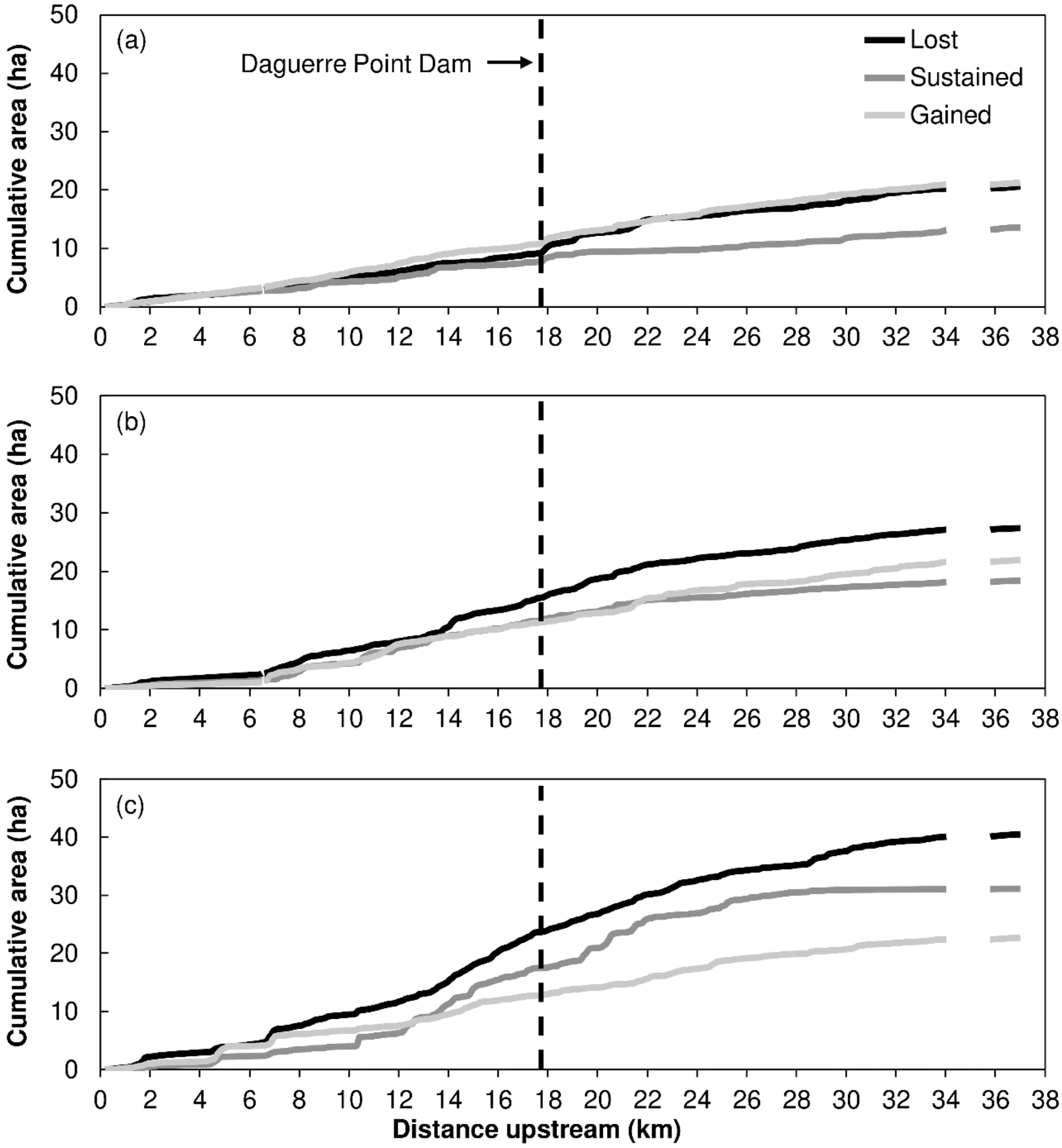

FIGURE 6 Longitudinal cumulative areas of preferred Chinook salmon fry rearing microhabitat lost, sustained, and gained between 2006/2008 and 2014 at (a) baseflow, (b) bankfull, and (c) flood discharges. The river was not modeled between river-km 34 and 36 due to its complex topography and hydraulics

Overall, areas of lost and gained microhabitat remained relatively even across habitat patch sizes and throughout the river segment at baseflow discharge. However, higher total areas of lost microhabitat patches, particularly between 10^2 and 10^3 m^2, caused an overall reduction in preferred microhabitat at the bankfull and flood discharges. Longitudinal cumulative areas indicate that the areas of lost microhabitat were relatively continuous throughout the river; however, the rate of increasing cumulative area of lost microhabitat appeared to be highest between river-km 14 and 22 at bankfull and flood discharges.

Changes in physical habitat conditions associated with lost, sustained, and gained microhabitat

Changes in channel alignment explained a significant but minor amount of the percentage change in preferred microhabitat. There was a 5.1, 4.9, and 0.6% decrease in wetted area between 2006/2008 and 2014 at baseflow, bankfull, and flood discharges, respectively. The percentage of habitat lost between 2006/2008 and 2014 as a result of preferred microhabitat becoming dry was 43, 38, and 18% at baseflow, bankfull, and flood discharges, respectively. The percentage of habitat gained from dry habitat becoming preferred microhabitat was 37, 38, and 41%, respectively.

In comparison to the effect of wetting and drying, the majority of habitat change is explained by systematic hydraulic changes. Specifically, areas where preferred microhabitat was lost tended to increase in depth and velocity, while areas where microhabitat was gained tended to decrease in depth and velocity (Table 1). Shifts to deeper and/or faster water may have caused microhabitats to become less suitable to Chinook salmon fry (as defined by the depth and velocity HSC functions used in this study), while shifts to shallower and/or slower water may have caused microhabitat to become more suitable. Areas where preferred microhabitat was sustained remained relatively constant in depth and velocity but did become slightly deeper and faster, on average.

TABLE 1 Mean and standard deviation of differences in depth and velocity between 2006/2008 and 2014 in lost, sustained, and gained Chinook salmon fry rearing preferred microhabitat. Positive values indicate increases in values, while negative values indicate decreases

Habitat condition	**Lost**	**Sustained**	**Gained**
Depth (m)			
Baseflow	0.36 ± 0.48	0.02 ± 0.27	-0.18 ± 0.51
Bankfull	0.43 ± 0.55	0.08 ± 0.24	-0.02 ± 0.40
Flood	0.49 ± 0.65	0.14 ± 0.23	0.12 ± 0.33
Velocity (m/s)			
Baseflow	0.19 ± 0.22	0.02 ± 0.09	-0.12 ± 0.30
Bankfull	0.30 ± 0.28	0.04 ± 0.11	-0.18 ± 0.36
Flood	0.36 ± 0.30	0.04 ± 0.10	-0.04 ± 0.23

The distribution of area of changes in depth and velocity showed similar results to Table 1 (Figure 7). Changes in depth tended to skew toward positive values (i.e., get deeper) for lost preferred microhabitat, remained centered around 0 for sustained microhabitat (i.e., small change), and were mixed for gained microhabitat. Changes in velocity followed a similar pattern skewing toward positive values for lost microhabitat, centered around 0 for sustained microhabitat, and mixed for gained microhabitat.

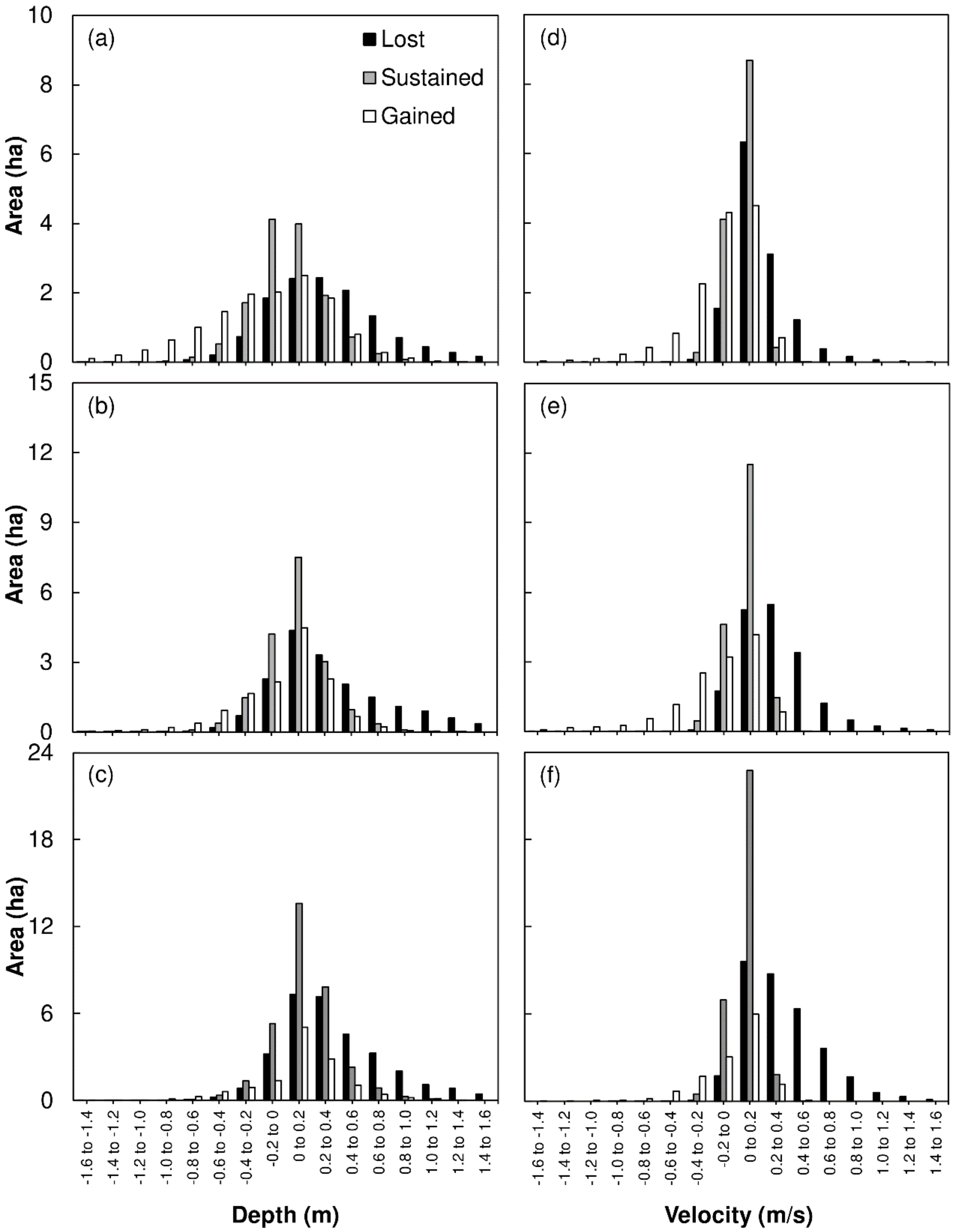

FIGURE 7 Areas associated with changes in depth between 2006/2008 and 2014 at (a) baseflow, (b) bankfull, and (c) flood discharges and with changes in velocity at (d) baseflow, (e) bankfull, and (f) flood discharges. Positive values indicate increases in depth and velocity, while negative values indicate a decrease

Percentages of lost, sustained, and gained preferred microhabitat area where there was a loss, gain, or no change in vegetative cover varied with discharge (Figure 8). At baseflow and flood discharges, there were relatively little differences in the percentages of area where vegetative cover was lost or gained within lost, sustained, and gained preferred microhabitat areas. Additionally, at those two discharges, at least 83% of the area within lost, sustained, and gained microhabitat areas had no change in cover. At bankfull discharge, however, the percentage of area where there was a gain in vegetative cover was over 3x higher in gained microhabitat areas compared to lost microhabitat areas. Furthermore, the percentage of area where there was a loss in vegetative cover was over 5x less in gained microhabitat areas compared to lost microhabitat areas. However, based on these results, changes in vegetative cover appeared to play a less significant role in explaining changes in habitat availability compared to changes in channel alignment and hydraulic conditions, particularly at baseflow and flood discharges.

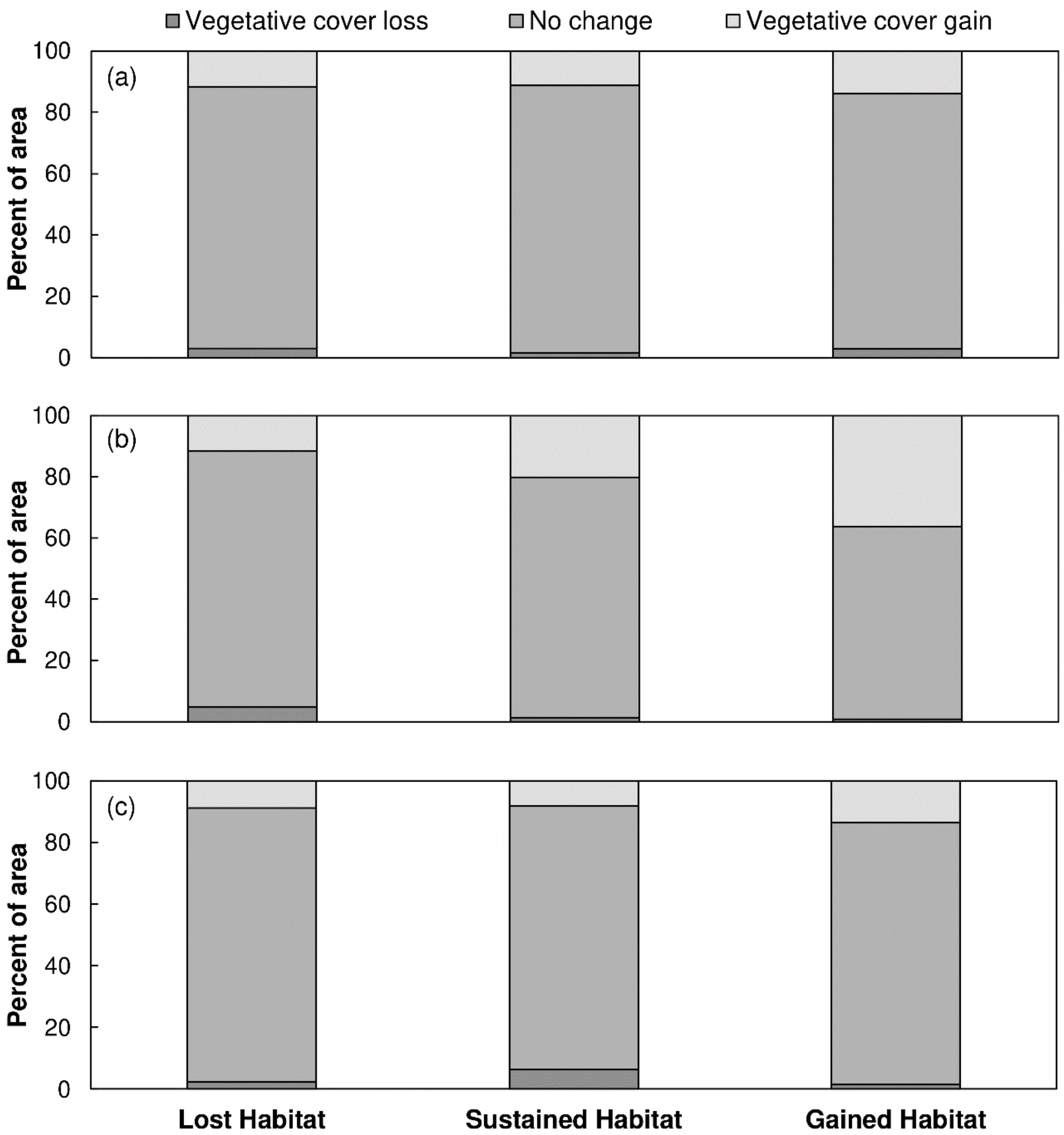

FIGURE 8 Percentage of lost, sustained, and gained microhabitat area where there was a loss, gain, or no change in vegetative cover at (a) baseflow, (b) bankfull, and (c) flood discharges

DISCUSSION

Calculating habitat availability

One of the most widely used metrics for deriving habitat availability from habitat suitability models is the weighted usable area (WUA) index, made popular by the physical habitat simulation model (PHABSIM; Bovee, 1986; Waddle, 2001) used in the Instream Flow Incremental Methodology (IFIM). This metric is calculated as the product of habitat area (i.e., cell size) and habitat suitability summed across a study domain (Bovee, 1986). Some studies have also nondimensionalized this index by dividing it by the area of the study domain (Benjankar et al., 2018; Mouton et al., 2007; Yao, Bui, & Rutschmann, 2018). Despite its widespread use, the WUA index has been highly criticized for lacking statistical certainty (Williams, 1996), interpretability (Mather, Bason, Purdy, & Silver, 1985), and biological meaning (Railsback, 2016). Unlike those indices, however, bioverified models such as the one used in this study provide biologically meaningful and spatially explicit interpretable areas of preferred microhabitat with known statistical certainty (Moniz et al., 2020). Furthermore, as demonstrated in this study, these areas of preferred microhabitat can be spatially compared over time to determine exactly where and how much habitat is lost, sustained and/or gained, and then compare those changes to changes in physical habitat conditions.

Comparing habitat change with geomorphic change

The period between 2006/2008 and 2014 was relatively dry in California, with the LYR only experiencing four modest ~3–5 year flood events of short duration, albeit with a brief peak discharge up to 9x bankfull. During this time period, the total area of in-channel Chinook salmon fry rearing habitat availability increased slightly, while overbank habitat decreased substantially. At the flood discharge specifically, there was a 25% decrease in preferred microhabitat area. Based on the results of this study, decreases in overbank habitat availability, particularly at the flood discharge, appeared to be associated with increases in depth and velocity (which resulted in diminished habitat suitability), primarily in patches between 10^2 and 10^3 m^2, and slightly more concentrated between river-km 14 and 22.

These results are consistent with findings from Weber and Pasternack (2017), who conducted topographic change detection analysis on the LYR spanning the same time period by calculating vertical differences in the 2006/2008 and 2014 DEMs. They found that erosion primarily occurred on floodplains, terraces, and lateral bars between river-km 14 and 22, while deposition occurred within the baseflow channel throughout the river segment. In other words, the channel and floodplain became better connected. Widespread erosion on the floodplain causing deeper and/or faster microhabitat conditions within the 2014 flood discharge wetted area may explain the overall reduction in habitat availability for Chinook salmon fry rearing observed in this study (Figure 9). However, at discharges <2x bankfull, Chinook salmon fry rearing habitat availability remained relatively consistent over this time period, despite significant bank erosion and lateral channel migration (Weber & Pasternack, 2017). These results suggest that despite being hydrogeomorphically dynamic, the LYR can self-sustain Chinook salmon fry rearing habitat during more frequently occurring, within-bankfull channel discharges. Additional research is needed to determine if these results are consistent during more significant flood events on the LYR. Although these specific results may not apply to all hydrologic periods on the LYR or for all rivers, the finding that habitat-discharge relationships can change relatively quickly as a result of morphodynamic processes likely does transfer globally and should be considered when analyzing habitat availability over decades.

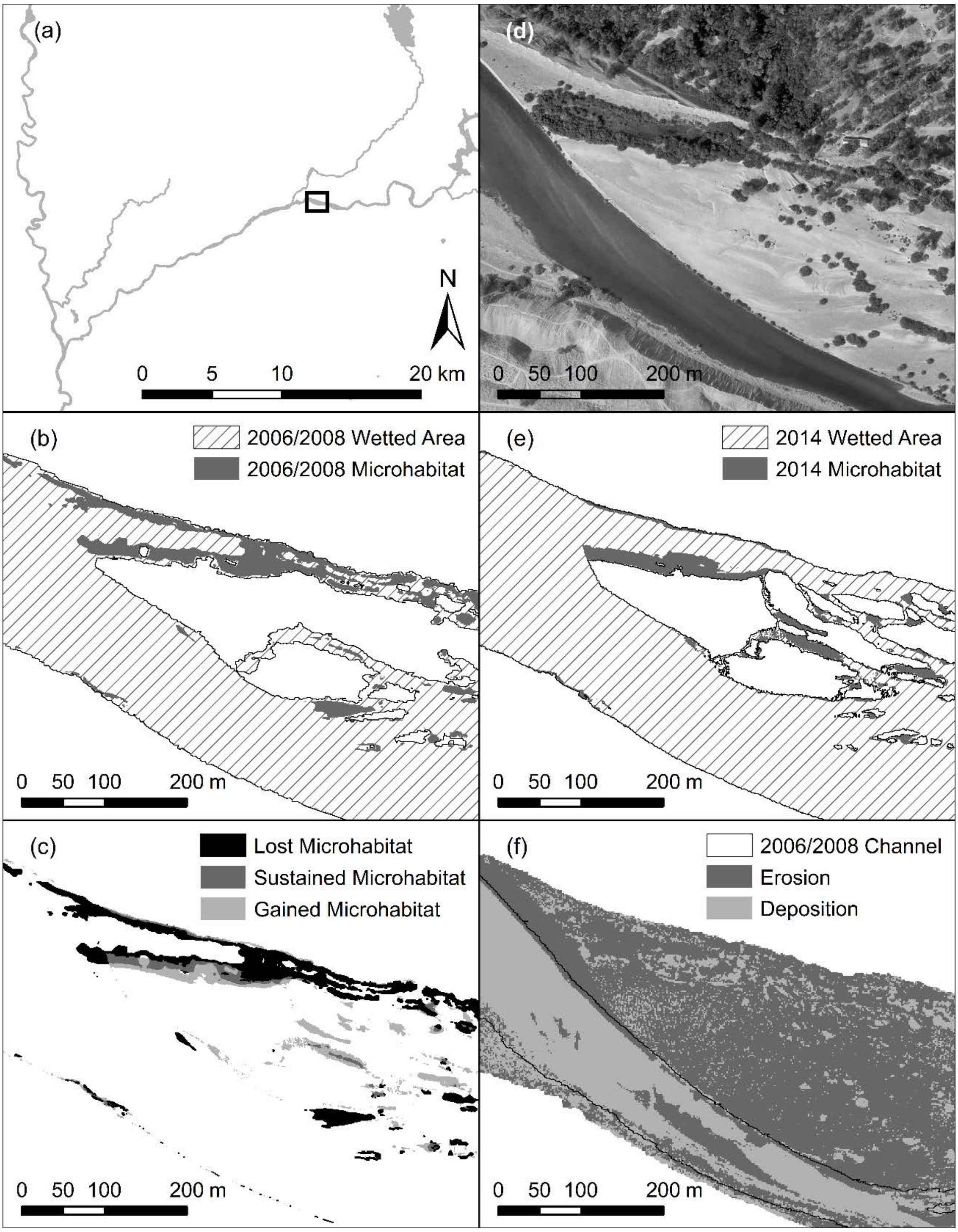

FIGURE 9 Example maps showing changes in preferred Chinook salmon fry rearing microhabitat and topography near river-km 22. (d) Aerial imagery was collected in 2008 and is shown for reference. (b, c, e) Wetted areas and microhabitat are shown at the flood discharge. (f) Raw topographic change results are shown with permission from Weber and Pasternack (2017)

Other studies have also attempted to relate changes in salmonid habitat suitability with geomorphic changes in rivers using repeat topographic surveys and habitat suitability models. Wheaton et al. (2010) found that Chinook salmon spawning habitat suitability remained relatively stable after gravel augmentation in locations where there were small elevation changes after a major flood event. Furthermore, they found that areas where habitat suitability degraded were dominated by larger magnitude erosion, whereas areas of habitat suitability improvement were dominated by shallow, low magnitude deposition. Harrison et al. (2011) qualitatively related changes in Chinook salmon spawning and rearing habitat availability with geomorphic changes in a restructured gravel-cobble river. They found that Chinook salmon spawning habitat availability increased over time, which they attributed to decreasing velocities over riffles. They also found that rearing habitat availability remained relatively low throughout the study period, which they attributed to high velocities in pools caused by flow constrictions by growing bars. Harrison et al. (2017) observed widespread deposition within the floodplain and erosion within the low-flow channel of a semiarid gravel-bed river after a 10-year flood event. Channel erosion roughly doubled the areal extent of pool mesohabitat, which they predicted would have a positive impact on steelhead trout (*O. mykiss*) habitat. Tamminga and Eaton (2018) observed a decline in the low-flow habitat suitability of adult and juvenile brown trout (*Salmo trutta*) after a 100-year flood event in a gravel-bed river in the Canadian Rocky Mountains. They attributed this reduction in habitat suitability to shallower flow conditions caused by bank erosion and in-channel deposition. Results from these studies and the study presented herein highlight the value of repeat surveys in the assessment of river ecosystem dynamics.

Metrics for evaluating aquatic habitat stability

As repeat topographic surveys and habitat modeling become more cost-efficient with increased availability of remote sensing data (Dimitriou & Stavroulaki, 2018; Tamminga & Eaton, 2018), the opportunity for resource managers to support these kinds of repeat studies will become more practical. Repeat studies can help resource managers better understand the dynamics and stability of rivers and their habitats at multiple spatial scales with varying hydrologic conditions, which can lead to more cost-effective long-term decision-making. Having a better understanding of aquatic habitat stability will become increasingly important as rivers continue to adjust to anthropogenic disturbances (Brown & Pasternack, 2017; Gregory, 2006; James, 1991; Leopold, 1973), potentially exacerbated by climate change (Hauer et al., 2013; Meybeck, 2003; Palmer et al., 2009; Praskievicz, 2015).

Yet, what constitutes the appropriate method and suite of metrics for evaluating aquatic habitat stability? This study attempts to answer that question with two different conceptual approaches. The first approach involves comparing a typical suite of bulk statistics (including minimums and maximums in the habitat-discharge relationship) for more than one time period, similar to what has been done in other studies addressing changes in habitat availability before and after restoration projects, dam removals, and extreme flood events. The second approach involves a new method in which lost, sustained, and gained microhabitat areas are spatially delineated and used to assess changes in specific physical habitat conditions. This new approach allows resource managers to better understand the locations and potential drivers of habitat change over time. In this case, it was possible to ascertain the size and spatial distribution of changes in Chinook salmon fry rearing habitat availability in the LYR, as well as the relative roles of channel alignment, hydraulic conditions, and vegetative cover in explaining these changes.

Although not examined in this study, spatially explicit areas of preferred microhabitat can also be evaluated for multiple species and/or multiple life stages of a single species of special concern. For example, this kind of analysis could be used by resource managers to help evaluate which discharges or restoration design alternatives reduce competition and/or predation of native species by nonnative species by minimizing areas of preferred microhabitat overlap. Furthermore, by overlapping preferred microhabitat for multiple life stages of a single species (e.g., salmonid spawning, embryo development, and rearing), managers could evaluate which discharge or design alternative maximized life stage connectivity throughout the year.

CONCLUSIONS

This study used new and existing methods to evaluate the stability of Chinook salmon fry rearing habitat availability in a regulated, yet dynamic, gravel-cobble river during a relatively mild hydrologic period. Results suggest that despite being hydrogeomorphically dynamic, the LYR can self-sustain Chinook salmon fry rearing habitat during more frequently occurring, within-bankfull channel discharges. At the floodplain-filling discharge, however, decreases in habitat availability were more prevalent, and appeared to be associated with increases in depth and velocity, particularly where widespread floodplain erosion had been previously observed. This study demonstrates the value of using electivity indices and repeat surveys to delineate areas of preferred habitat stability, and highlights the importance of considering morphodynamics in aquatic habitat analysis, even during relatively mild hydrologic periods. As aquatic ecosystems continue to adjust to both natural and anthropogenic disturbances, this consideration will become increasingly important in guiding long-term regulatory and management decisions, such as instream flow requirements and habitat restoration plans.

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CONFLICT OF INTEREST

The authors declare that there is no conflict of interest that could be perceived as prejudicing the impartiality of the research reported.

DATA AVAILABILITY STATEMENT

The data presented in tables and figures that support the findings of this study are available from the senior author (http://pasternack.ucdavis.edu) upon request with no restrictions. All 2006/2008 DEM, hydrodynamic models, and microhabitat models/data are available publicly through the Federal Energy Regulatory Commission. Restrictions apply to the availability of the 2014 DEM, hydrodynamic models, and microhabitat models/data, as these were used under contractual agreement from the project sponsor. These are available from the senior author with the permission of Yuba Water Agency.

REFERENCES

Abu-Aly, T. R., Pasternack, G. B., Wyrick, J. R., Barker, R., Massa, D., & Johnson, T. (2014). Effects of LiDAR-derived, spatially distributed vegetation roughness on two-dimensional hydraulics in a gravel-cobble river at flows of 0.2 to 20 times bankfull. *Geomorphology*, *206*, 468–482. https://doi.org/10.1016/j.geomorph.2013.10.017

Ahmadi-Nedushan, B., St-Hilaire, A., Bérubé, M., Robichaud, É., Thiémonge, N., & Bobée, B. (2006). A review of statistical methods for the evaluation of aquatic habitat suitability for instream flow assessment. *River Research and Applications*, *22*(5), 503–523. https://doi.org/10.1002/rra.918

Barker, J., Pasternack, G. B., Bratovich, P., Duane, M., Wyrick, J. R., & Johnson, T. (2018). Kayak drifter surface velocity observation for 2D hydraulic model validation. *River Research and Applications*, *34*(2), 124–134. https://doi.org/10.1002/rra.3238

Beecher, H. A., Carleton, J. P., & Johnson, T. H. (1995). Notes: Utility of depth and velocity preferences for predicting steelhead parr distribution at different flows. *Transactions of the American Fisheries Society*, *124*, 935–938. https://doi.org/10.1577/1548-8659(1995)124<0935:NUODAV>2.3.CO;2

Benjankar, R., Tonina, D., McKean, J. A., Sohrabi, M. M., Chen, Q., & Vidergar, D. (2018). Dam operations may improve aquatic habitat and offset negative effects of climate change. *Journal of Environmental Management*, *213*, 126–134. https://doi.org/10.1016/j.jenvman.2018.02.066

Boavida, I., Caetano, L., & Pinheiro, A. N. (2020). E-flows to reduce the hydropeaking impacts on the Iberian barbel (*Luciobarbus bocagei*) habitat. An effectiveness assessment based on the COSH Tool application. *Science of the Total Environment*, *699*, 1–12. https://doi.org/10.1016/j.scitotenv.2019.134209

Bovee, K. D. (1986). *Development and* evaluation *of habitat suitability criteria for use in the instream flow incremental methodology*. Washington, DC: US Fish and Wildlife Service. Instream Flow Information Paper #21 FWS/OBS-86/7.

Brown, R. A., & Pasternack, G. B. (2017). Bed and width oscillations form coherent patterns in a partially confined, regulated gravel–cobble-bedded river adjusting to anthropogenic disturbances. *Earth Surface Dynamics*, *5*(1), 1–20. https://doi.org/10.5194/esurf-5-1-2017

Burman, S. G., & Pasternack, G. B. (2017). *Riparian canopy abundance, distribution, and height on the lower Yuba River in 2008*. Davis, CA: University of California at Davis. Prepared for the Yuba Accord River Management Team.

Carley, J. K., Pasternack, G. B., Wyrick, J. R., Barker, J. R., Bratovich, P. M., Massa, D. A., Reedy, G. D., Johnson, T. R. (2012). Significant decadal channel change 58–67 years post-dam accounting for uncertainty in topographic change detection between contour maps and point cloud models. *Geomorphology*, *179*, 71–88. https://doi.org/10.1016/j.geomorph.2012.08.001

Dimitriou, E., & Stavroulaki, E. (2018). Assessment of riverine morphology and habitat regime using unmanned aerial vehicles in a Mediterranean Environment. *Pure and Applied Geophysics*, *175*, 3247–3261. https://doi.org/10.1007/s00024-018-1929-3

Dunbar, M. J., Alfredsen, K., & Harby, A. (2012). Hydraulic-habitat modelling for setting environmental river flow needs for salmonids. *Fisheries Management and Ecology*, *19*(6), 500–517. https://doi.org/10.1111/j.1365-2400.2011.00825.x

ESRI. (2018). *ArcGIS Desktop: Release 10.6*. Redlands, CA: Environmental Systems Research Institute.

Gard, M. (2006). Modeling changes in salmon spawning and rearing habitat associated with river channel restoration. *International Journal of River Basin Management*, *4*(3), 201–211. https://doi.org/10.1080/15715124.2006.9635289

Gard, M. (2014). Modelling changes in salmon habitat associated with river channel restoration and flow-induced channel alternations. *River Research and Applications*, *30*(1), 40–44. https://doi.org/10.1002/rra.2642

Gilbert, G. K. (1917). *Hydraulic-mining debris in the Sierra Nevada*. Washington, DC: US Geological Survey.

Gregory, K. J. (2006). The human role in changing river channels. *Geomorphology*, *79*(3–4), 172–191. https://doi.org/10.1016/j.geomorph.2006.06.018

Harrison, L. R., Legleiter, C. J., Wydzga, M. A., & Dunne, T. (2011). Channel dynamics and habitat development in a meandering gravel bed river. *Water Resources Research*, *47*, W04513. https://doi.org/10.1029/2009WR008926

Harrison, L. R., Pike, A., & Boughton, D. A. (2017). Coupled geomorphic and habitat response to a flood pulse revealed by remote sensing. *Ecohydrology*, *10*(5), 1–13. https://doi.org/10.1002/eco.1845

Hauer, C., Unfer, G., Holzmann, H., Schmutz, S., & Habersack, H. (2013). The impact of discharge change on physical instream habitats and its response to river morphology. *Climate Change*, *116*(3–4), 827–850. https://doi.org/10.1007/s10584-012-0507-4

Hopkins, C. E., & Pasternack, G. B. (2018). *Autumn 2014 lower Yuba River TUFLOW GPU 2D model description and validation*. Davis, CA: University of California at Davis. Prepared for Yuba County Water Agency.

Im, D., Kang, H., Kim, K., & Choi, S. (2011). Changes of river morphology and physical fish habitat following weir removal. *Ecological Engineering*, *37*(6), 883–892. https://doi.org/10.1016/j.ecoleng.2011.01.005

Jackson, J. R., Pasternack, G. B., & Wyrick, J. R. (2013). *Substrate of the lower Yuba River*. Davis, CA: University of California at Davis. Prepared for the Yuba Accord River Management Team.

James, L. A. (1991). Incision and morphologic evolution of an alluvial channel recovering from hydraulic mining sediment. *Geological Society of America Bulletin*, *103*(6), 723–736. https://doi.org/10.1130/0016-7606(1991)103<0723:IAMEOA>2.3.CO;2

James, L. A. (2005). Sediment from hydraulic mining detained by Englebright Dam and small dams in the Yuba Basin. *Geomorphology*, *71*(1–2), 202–226. https://doi.org/10.1016/j.geomorph.2004.02.016

Leopold, L. B. (1973). River channel change with time: An example. *Geological Society of America Bulletin*, *84*(6), 1845–1860. https://doi.org/10.1130/0016-7606(1973)84<1845:RCCWTA>2.0.CO;2

Mandlburger, G., Hauer, C., Wieser, M., & Pfeifer, N. (2015). Topo-bathymetric LiDAR for monitoring river morphodynamics and instream habitats – a case study at the Pielach River. *Remote Sensing*, *7*(5), 6160–6195. https://doi.org/10.3390/rs70506160

Mather, D., Bason, W. H., Purdy, E. J., & Silver, C. A. (1985). A critique of the instream flow incremental methodology. *Canadian Journal of Fisheries and Aquatic Sciences*, *42*(4), 825–831. https://doi.org/10.1139/f85-105

McMahon, T. E., & Hartman, G. F. (1989). Influence of cover complexity and current velocity on winter habitat by juvenile coho salmon (*Oncorhynchus kisutch*). *Canadian Journal of Fisheries and Aquatic Science*, *46*, 1551–1557. https://doi.org/10.1139/f89-197

Meybeck, M. (2003). Global analysis of river systems: From earth system controls to Anthropocene syndromes. *Philosophical Transactions of the Royal Society B*, 358(1440), 1935–1955. https://doi.org/10.1098/rstb.2003.1379

Moniz, P. J., Pasternack, G. B., Massa, D. A., Stearman, L. W., & Bratovich, P. M. (2020). Do rearing salmonids predictably occupy physical microhabitat? *Journal of Ecohydraulics*, *5*(2), 132–150. https://doi.org/10.1080/24705357.2019.1696717

Mouton, A. M., Schneider, M., Depestele, J., Goethals, P. L. M., & De Pauw, N. (2007). Fish habitat modelling as a tool for river management. *Ecological Engineering*, *29*(3), 305–315. https://doi.org/10.1016/j.ecoleng.2006.11.002

Naman, S. M., Rosenfeld, J. S., Neuswanger, J. R., Enders, E. C., & Eaton, B. C. (2019). Comparing correlative and bioenergetics-based habitat suitability models for drift-feeding fishes. *Freshwater Biology*, *64*(9), 1613–1626. https://doi.org/10.1111/fwb.13358

National Marine Fisheries Service. (2014). *Recovery plan for the evolutionarily significant units of Sacramento River winter-run Chinook salmon and Central Valley spring-run Chinook salmon and the distinct population segment of California Central Valley steelhead*. Sacramento, CA: NMFS Sacramento Protected Resources Division.

Palmer, M. A., Lettenmaier, D. P., Poff, N. L., Postel, S. L., Richter, B., & Warner, R. (2009). Climate change and river ecosystems: protection and adaptation options. *Environmental Management*, *44*, 1053–1068. https://doi.org/10.1007/s00267-009-9329-1

Pasternack, G. B. (2011). *2D modeling and ecohydraulic analysis*. Createspace: Seattle, WA.

Pasternack, G. B., & Hopkins, C. E. (2017). *Near-census 2D model comparison between SRH-2D and TUFLOW GPU for use in gravel/cobble rivers*. Davis, CA: University of California at Davis. Prepared for Yuba County Water Agency.

Pasternack, G. B., & Wyrick, J. R. (2017). Flood-driven topographic changes in a gravel-cobble river over segment, reach, and morphological unit scales. *Earth Surface Processes and Landforms, 42*(3), 487–502. http://dx.doi.org/10.1002/esp.4064

Praskievicz, S. (2015). A coupled hierarchical modeling approach to simulating the geomorphic response of river systems to anthropogenic climate change. *Earth Surface Processes and Landforms*, *40*(12), 1616–1630. https://doi.org/10.1002/esp.3740

Railsback, S. F. (2016). Why it is time to put PHABSIM out to pasture. *Fisheries*, *41*(10), 720–725. https://doi.org/10.1080/03632415.2017.1380985

Rosenfeld, J., Beecher, H., & Ptolemy, R. (2016). Developing bioenergetics-based habitat suitability curves for instream flow models. *North American Journal of Fisheries Management*, *36*(5), 1205–1219. https://doi.org/10.1080/02755947.2016.1198285

Sawyer, A. M., Pasternack, G. B., Moir, H. J., & Fulton, A. A. (2010). Riffle-pool maintenance and flow convergence routing confirmed on a large gravel bed river. *Geomorphology*, *114*, 143–160. https://doi.org/10.1016/j.geomorph.2009.06.021

Shirvell, C. S. (1990). Role of instream rootwads as juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*O. mykiss*) cover habitat under varying streamflows. *Canadian Journal of Fisheries and Aquatic Science*, *47*, 852–861. https://doi.org/10.1139/f90-098

Shirvell, C. S. (1994). Effect of changes in streamflow on the microhabitat use and movements of sympatric juvenile coho (*Oncorhynchus kisutch*) and Chinook salmon (*O. tshawytscha*) in a natural stream. *Canadian Journal of Fisheries and Aquatic Science, 51*, 1644–1652. https://doi.org/10.1139/f94-165

Stamou, A., Polydera, A., Papadonikolaki, G., Martínez-Capel, F., Muñoz-Mas, R., Papadaki, Ch., Zogaris, S., Bui, M.-D., Rutschmann, P., & Dimitriou, E. (2018). Determination of environmental flows in rivers using an integrated hydrological-hydrodynamic-habitat modelling approach. *Journal of Environmental Management*, *209*, 273–285. https://doi.org/10.1016/j.jenvman.2017.12.038

Tamminga, A., & Eaton, B. (2018). Linking geomorphic change due to floods to spatial hydraulic habitat dynamics. Ecohydrology, 11(8), 1–17. https://doi.org/10.1002/eco.2018

Tomsic, C. A., Granata, T. C., Murphy, R. P., & Livchak, C. J. (2007). Using a coupled eco-hydrodynamic model to predict habitat for target species following dam removal. *Ecological Engineering*, *30*(3), 215–230. https://doi.org/10.1016/j.ecoleng.2006.11.006

US Fish and Wildlife Service. (2010). *Flow-habitat relationships for juvenile fall/spring-run Chinook salmon and steelhead/rainbow trout rearing in the Yuba River*. Sacramento, CA: US Fish and Wildlife Service.

Vaughan, M. C., & Pasternack, G. B. (2014). *Aerial mapping of streamwood and human-built detritus available as cover in the lower Yuba River in autumn 2008*. Davis, CA: University of California at Davis. Prepared for the Yuba Accord River Management Team.

Waddle, T. J. (2001). PHABSIM *for Windows: User's manual and exercises*. Fort Collins, CO: US Geological Survey. Open File Report 2001-340.

Weber, M. D., & Pasternack, G. B. (2016). *2014 topographic mapping of the lower Yuba River*. Davis, CA: University of California at Davis. Prepared for the Yuba Accord River Management Team.

Weber, M. D., & Pasternack, G. B. (2017). Valley-scale morphology drives differences in fluvial sediment budgets and incision rates during contrasting flow regimes. *Geomorphology*, *288*, 39–51. https://doi.org/10.1016/j.geomorph.2017.03.018

Wheaton, J. M., Brasinton, J., Darby, S. E., Merz, J., Pasternack, G. B., Sear, D., & Vericat, D. (2010). Linking geomorphic changes to salmonid habitat at a scale relevant to fish. *River Research and Applications*, *26*(4), 469–486. https://doi.org/10.1002/rra.1305

Williams, J. G. (1996). Lost in space: Minimum confidence intervals for idealized PHABSIM studies. *Transactions of the American Fisheries Society*, *125*(3), 458–465. https://doi.org/10.1577/1548-8659(1996)125<0458:LISMCI>2.3.CO;2

Wyrick, J. R., & Pasternack, G. B. (2012). *Landforms of the lower Yuba River*. Davis, CA: University of California at Davis. Prepared for the Yuba Accord River Management Team.

Wyrick, J. R., & Pasternack, G. B. (2014). Geospatial organization of fluvial landforms in a gravel-cobble river: Beyond the riffle-pool couplet. *Geomorphology*, *213*, 48–65. https://doi.org/10.1016/j.geomorph.2013.12.040

Wyrick, J. R., & Pasternack, G. B. (2015). Revealing the natural complexity of topographic change processes through repeat surveys and decision-tree classification. *Earth Surf Processes and Landforms*, *41*(6), 723–737. https://doi.org/10.1002/esp.3854

Yao, W., Bui, M. D., & Rutschmann, P. (2018). Development of eco-hydraulic model for assessing fish habitat and population status in freshwater ecosystems. *Ecohydrology*, *11*(5), 1–17. https://doi.org/10.1002/eco.1961

Yuba Water County Agency. (2012). *Model Report. Appendix A. Hydrology Report (No. Yuba River Development Project FERC Project No. 2246)*. Marysville, CA: Yuba County Water Agency.

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