

Title:

Upscaling Site-Scale Ecohydraulic Models to Inform Salmonid Population-Level Life Cycle Modelling and Restoration Actions – Lessons from the Columbia River Basin

Short Title:

Upscaling Ecohydraulic Models

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Manuscript Description:

New insights into what quantifying instream fish habitat can tell us about the populations of threatened salmon.

This is the author manuscript accepted for publication and has undergone full peer review but has not been through the copyediting, typesetting, pagination and proofreading process, which may lead to differences between this version and the Version of Record. Please cite this article as doi: [10.1002/esp.4137](https://doi.org/10.1002/esp.4137)

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For Submission to Earth Surface Processes and Landforms as Research Article for consideration in Special Issue: 'Remote sensing applications for hydro and morphodynamic monitoring & modelling'

## **Abstract**

With high-resolution topography and imagery in fluvial environments, the potential to quantify physical fish habitat at the reach-scale has never been better. Increased availability of hydraulic, temperature and food availability data and models have given rise to a host of species and life stage specific ecohydraulic fish habitat models ranging from simple, empirical habitat suitability curve driven models, to fuzzy inference systems to fully mechanistic bioenergetic models. However, few examples exist where such information has been upscaled appropriately to evaluate entire fish populations. We present a framework for applying such ecohydraulic models from over 905 sites in 12 sub-watersheds of the Columbia River Basin (USA), to assess status and trends in anadromous salmon populations. We automated the simulation of computational engines to drive the hydraulics, and subsequent ecohydraulic models using cloud computing for over 2075 visits from 2011 to 2015 at 905 sites. We also characterize each site's geomorphic reach type, habitat condition, geomorphic unit assemblage, primary production potential and thermal regime. We then independently produce drainage network-scale models to estimate these same parameters from coarser, remotely sensed data available across entire populations within the Columbia River Basin. These variables give us a basis for imputation of reach-scale capacity estimates across drainage networks. Combining capacity estimates with survival estimates from mark-recapture monitoring allows a more robust quantification of capacity for freshwater life stages (i.e. adult spawning, juvenile rearing) of the anadromous lifecycle. We use these data to drive life cycle models of populations, which not only include the freshwater life stages but also the marine and migration life stages through the hydropower system. More fundamentally, we can begin to look at more realistic, spatially explicit, tributary habitat restoration scenarios to examine whether the enormous financial investment on such restoration actions can help recover these populations or prevent their extinction.

**Keywords:**

salmon recovery, stream and river restoration, *Oncorhynchus* sp., fish habitat, population modelling

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## Introduction

Globally, native anadromous salmonid populations are in decline and several face the threat of extinction (Gustafson *et al.*, 2007; Rand *et al.*, 2012). Given the cultural, economic, and ecological importance of these populations, numerous efforts are underway to mitigate impacts that lead to their decline and facilitate their recovery (e.g. conservation hatcheries, (Maynard and Trial, 2013), habitat restoration, (Barnas *et al.*, 2015)). For example, in the Columbia River Basin, there are eleven evolutionarily significant units of salmon or steelhead (*Oncorhynchus* sp.) listed as threatened and two as endangered under the U.S. Endangered Species Act (ESA) (Ruckelshaus *et al.*, 2002; Gustafson *et al.*, 2007). According to the ESA review on the risks posed to anadromous salmonids by the Columbia Basin's federally administered hydropower dam system ((NOAA), 2008), three action agencies are required to make costly dam retrofits and operational changes (i.e., to improve passage) and/or mitigate dam impacts via offsite actions (i.e., tributary habitat restoration). The Bonneville Power Administration's Tributary Habitat Improvement Program alone spends US\$100-150M annually on habitat recovery efforts and another US\$90M on research, monitoring and evaluation projects ((NPCC), 2015; Leonard *et al.*, 2015). Arguably, the advancement of ecohydraulics owes much of its existence and history to applied science efforts like this. Such programs have spurred the development of new methods for understanding the nature of habitat impacts on fish. These methods are used for assessing the status and trends of fish and habitat, as well as the effectiveness of restoration actions. The field of ecohydraulics, particularly as informed by fluvial geomorphology, has helped better define, understand, model and predict the importance of physical habitat to fish (Pasternack, 2011).



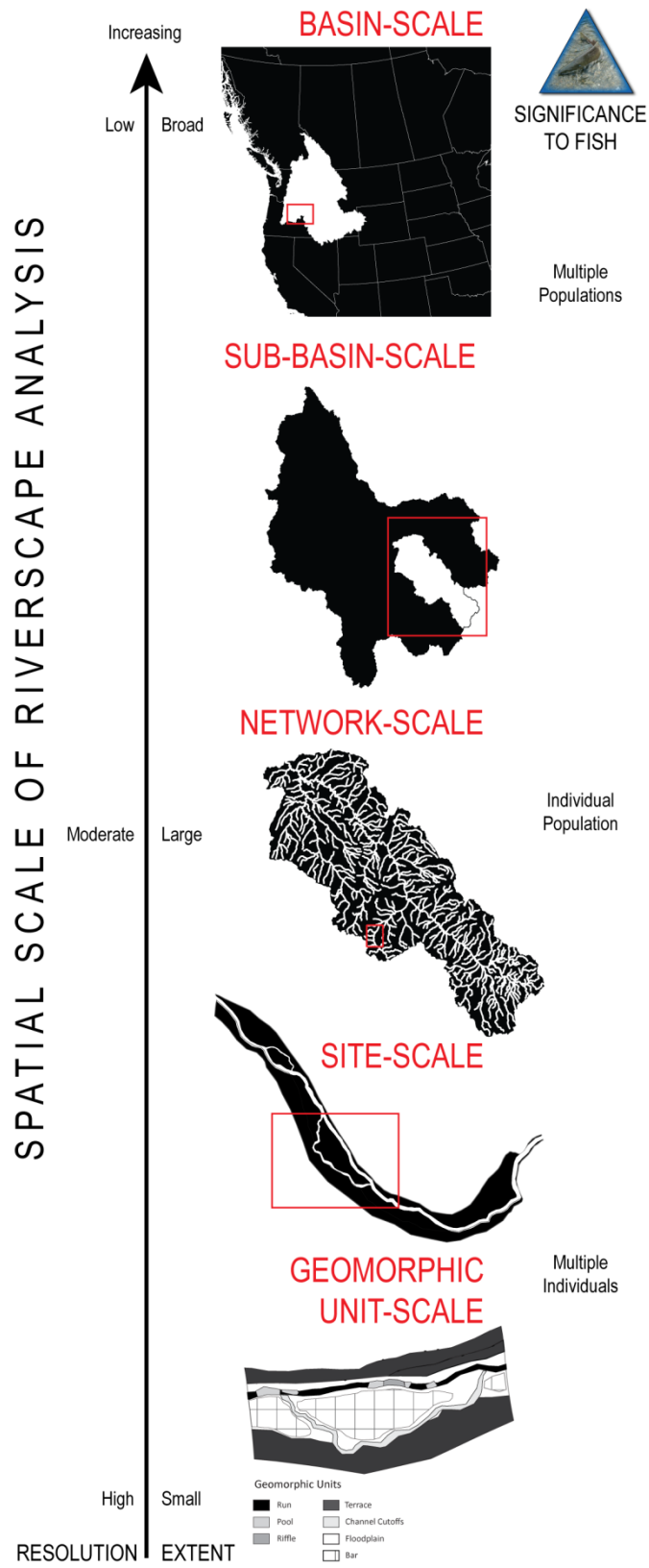


Figure 1 - Illustration of hierarchical spatial scales for considering habitat. See Table 1 for definitions.

Table 1 – Definition of spatial scales used in this paper in terms of extent and resolution.

Scale Name	Extent	Resolution	GIS Representation
Basin-Scale	Basin ( $10^3$ to $10^5$ km <sup>2</sup> )	Basin ( $10^3$ to $10^5$ km <sup>2</sup> )	Polygon
Sub-Basin-Scale	Sub-Basin ( $10^1$ to $10^3$ km <sup>2</sup> )	Sub-Basin ( $10^1$ to $10^3$ km <sup>2</sup> )	Polygon
Network-Scale	Sub-Basin (Areal: $10^1$ to $10^3$ km <sup>2</sup> ; Length: $10^4$ to $10^8$ m)	Reach ( $10^2$ to $10^3$ m)	Polyline
Reach-Scale	Reach ( $10^2$ to $10^3$ m)	Geomorphic Unit ( $10^0$ to $10^2$ m)	Raster or Polygon
Sites-on-Network-Scale	Sub-Basin ( $10^1$ to $10^3$ km <sup>2</sup> )	Site Locations (points)	Point
Site-Summary-Scale	Reach ( $10^2$ to $10^3$ m)	Reach ( $10^2$ to $10^3$ m)	Point, Polyline or Polygon
Site-Scale (CHaMP Reach)	Reach (20 x Bankfull width; 160 m to 600 m)	Cell ( $10^{-1}$ to $10^0$ m)	Raster
Geomorphic-Unit-Scale	Geomorphic Unit ( $10^0$ to $10^2$ m)	Hydraulic Unit ( $10^0$ to $10^1$ m) or Cell	Polygon
Hydraulic-Unit-Scale	Hydraulic Unit ( $10^0$ to $10^1$ m)	Grain ( $10^{-2}$ to $10^0$ m)	Polygon
Cell-Scale	Cell ( $10^{-1}$ to $10^0$ m)	Cell ( $10^{-1}$ to $10^0$ m)	Raster

Advances in our ability to map riverscapes with aerial imagery (e.g. Gilvear *et al.*, 2007; Legleiter *et al.*, 2009) and a multitude of new topographic survey techniques (Passalacqua *et al.*, 2015) across a range of spatial scales have transformed the ways in which rivers can be read and interpreted (McKean *et al.*, 2009; Fonstad and Marcus, 2010; Carbonneau *et al.*, 2011; Carbonneau and Piegay, 2012). These techniques have matured and revolutionized what is possible to resolve and quantify with respect to fish habitat at a reach scale (see Table 1 and Figure 1 for scale definitions in terms of resolution and extent). Recent reviews by Passalacqua *et al.* (2015), Tarolli (2014) and Harpold *et al.* (2015) synthesize advances in acquisition of high resolution topography (i.e. < 2 m resolution), as well as imagery. In some environments, certain technologies (e.g. green lidar, multibeam sonar, structure from motion, spectral-depth correlation) can even provide near-continuous, high resolution coverage at ‘segment-scale’ extents, thereby providing detailed insight at an extent that Fausch *et al.*

(2002) argued was critically missing almost 15 years ago. Many of the ecohydraulic advances over the past two decades are the direct result of improved remote sensing capabilities – the topic of this special issue and the related session at the 11<sup>th</sup> International Symposium on Ecohydraulics.

With a few exceptions across limited spatial extents (e.g. McKean *et al.*, 2009; Fonstad and Marcus, 2010; Carnie *et al.*, 2016), none of these technologies can yet deliver the promise of a habitat census across the diversity of riverscapes occupied by salmonids (Bangen *et al.*, 2014a). While hybrid approaches combining multiple techniques have emerged as the most realistic way to obtain complete coverage of a portion of a riverscape (e.g. Legleiter, 2012; Javernick *et al.*, 2014; Williams *et al.*, 2014), for now, the notion of a physical habitat census is something we can only achieve across limited portions (albeit still impressive at 10-50 km) of rivers (e.g. Pasternack, 2011; Grams *et al.*, 2013; Wyrick and Pasternack, 2014; Benjankar *et al.*, 2016). Wyrick and Pasternack (2012) coined the term ‘near-census’ to describe impressive efforts to collect high-resolution topography and imagery, continuously at the riverscape-scale. Yet, detailed ‘near-census’ mapping of physical habitat assessments continue to elude network-scale ambitions (i.e. 100’s to 1000’s of kilometers of stream) in all but the smallest of watersheds (e.g. McMillan *et al.*, 2013), thereby necessitating sampling-based approaches in most systems.

Beyond questions of spatial scale, physical habitat data must be translated into a currency that can inform status assessments for populations of anadromous salmonids (Isaak *et al.*, 2016). Estimates of carrying capacity and productivity are commonly used in salmonid life-cycle modeling to estimate likelihood for population persistence and recovery potential (Moussalli and Hilborn, 1986). Capacity for a given life stage refers to the upper limit or maximum number of fish that any spatial extent of a riverscape (see Table 1) can support (e.g. Beechie *et al.*, 2006). Productivity represents the maximum survival between life stages, which can be estimated from fish monitoring efforts (e.g. Thorson *et al.*, 2014). Anadromous salmonids begin their life cycle as eggs in the gravel beds of rivers, continue to grow and feed as juveniles in freshwater habitats, and then migrate (sometimes over 1000s of kilometers) to the ocean, where they spend 2-4 years feeding and growing; upon reaching maturity, they then reenter freshwater and return to their natal spawning grounds to spawn (Groot and Margolis,

1991). Juvenile and adult capacity refers to the maximum number of juveniles the habitat can support and the number of redds spawners could successfully build, respectively. While this is simple conceptually, habitat data are rarely used to inform salmon population assessments in a robust, meaningful way (Sweka and Wainwright, 2013).

Despite the ever increasing resolution and extent of data available to drive traditional ecohydraulic models (e.g. Leclerc *et al.*, 1995; Pasternack *et al.*, 2004; Leclerc, 2005; Mouton *et al.*, 2007), the ability of these data/tools to address questions of how habitat conditions will result in population-level responses remains unclear (Sweka and Wainwright, 2013). This may be a result of i) a lack of habitat data at a scale or extent that matters to a population, ii) the use of study designs that fail to capture the huge variability in fish habitat that influences a population (Larsen *et al.*, 2001), and/or iii) a lack of jurisdictional or institutional coordination between governmental entities monitoring fish populations (e.g. state fish and wildlife agencies) and/or habitats (e.g. federal water, energy and land management agencies). Further, because the life history of salmon and steelhead also exposes them to anthropogenic threats outside of natal streams, rigorous studies on how geomorphology and ecohydraulics influence these fish may not reveal a basis for population limitation or recovery (Budy and Schaller, 2007). The challenge is how to objectively determine the extent to which habitat may matter and whether improvements through restoration could help increase fish production (i.e. the product of survival, growth and abundance).

The purpose of this paper is to describe a framework that leverages field data (i.e., high-resolution topography, and other habitat variables) to drive reach-scale ecohydraulic models that, following an upscaling step (i.e., from site/reach to network/population using remote sensing data), can inform population-level assessments of ESA-listed salmonids. Specifically, we illustrate this framework with the Columbia Habitat Monitoring Program (CHaMP) that aims to quantify the capacity of tributary streams to support both juvenile and adult spawning life stages of priority salmon populations. CHaMP achieves this by drawing on a host of technological advances and research discoveries in the fields of ecohydraulics, fluvial geomorphology, fish ecology, remote sensing and computer science. The program conducts surveys at over 600 sites per year throughout the Columbia River Basin, and thus produces an enormous amount of reach-scale data. Subsequently, the program has had to build and

operationalize a number of tools and existing analytical techniques into a framework to address key management questions relevant to salmon recovery. We do not cover the framework's myriad of component pieces or specific advances in remote sensing that facilitate this integration in great detail. Although, we do refer the reader to relevant examples from the literature where appropriate. Instead, we sketch for the reader a well-integrated, thoughtfully designed framework. Although we specifically developed the framework for Pacific salmon in the Columbia Basin, the framework has relevance to virtually any salmonid with complex habitat associations and life history expression at multiple scales. The paper should help geomorphologists see some of the ways in which their research could inform more than simply better understanding fish habitat, but also make direct connections to implications for entire fish populations.

### **Key Management Questions & Role of Research, Monitoring and Evaluation in Columbia River Basin**

We will use the case study of managing anadromous salmonid stocks from the Columbia River Basin (CRB), in the Pacific Northwest of the United States to explore the theme of leveraging remote sensing advances across a range of scales to develop an ecohydraulic and geomorphic informed assessment of the impacts of fish habitat condition on fish populations. The principles and approaches described here have utility beyond the CRB. The causes of population decline and need for recovery are by no means unique to this basin nor salmonids alone (Dudgeon *et al.*, 2006). These anthropogenic impacts are typically summarized in terms of the four H's – Harvest, Hatcheries, Hydropower and Habitat, which individually act to limit survival of natural stocks at specific life stages and collectively limit production (Ruckelshaus *et al.*, 2002). In the CRB, more than 300 dams, including 25+ on the mainstem Columbia and Snake Rivers, directly (e.g. passage) or indirectly (e.g. river hydrology and temperature) impact anadromous salmonids (Leonard *et al.*, 2015). Accordingly, considerable investments have been made (and are still ongoing) by dam operators (U.S. Army Corps of Engineers and Bureau of Reclamation) and hydropower marketers (Bonneville Power Administration) to improve passage and survival for both out-migrating juveniles and adults returning to spawn. However, the overarching recovery strategy relies heavily on restoration actions aimed at improving salmonid survival and production in their tributary life stages,

predominantly the egg to smolt stages (Kareiva *et al.*, 2000). More specifically, restoration aims to create productivity or capacity gains by offsetting (or reversing) habitat degradation from a myriad of impactful historic and contemporary land use practices, such as timber harvest, extractive mining, arable agriculture, over-grazing, extirpation of beaver, and water diversions, among others (Meehan, 1991).

Under NOAA's 2008 Biological Opinion ((NOAA), 2008), a series of 'reasonable and prudent alternatives' (RPAs) to removing the mainstem dams of the hydropower system were identified (FCRPS, 2008). From FCRPS (2008), RPAs 56 and 57 identify tributary habitat restoration as a key, and required restoration component. BPA alone spends in excess of US\$100M a year on such actions, and invests another c. US\$90M a year on research, monitoring and evaluation (RM&E) culminating in well over 23,000 restoration projects to date at over 35,000 locations throughout the CRB between 1991 and 2005 (Katz *et al.*, 2007). The efforts in the CRB dwarf the scope of restoration identified in the heavily cited Bernhardt *et al.* (2005) paper, which reported the scope of efforts from a database across the US, and identified roughly 37,000 projects. The CRB is a massive system covering 668,000 km<sup>2</sup> (for comparison, France is 643,000 km<sup>2</sup>), with over 288,011 km of perennially flowing rivers and streams, 36,348 km (13%) and 19,182 km (7%) of which are still capable of supporting Chinook salmon and steelhead runs, respectively (Figure 2).

Three key management questions (Figure 3) emerge from RPAs 56 and 57, which represent both scientific and prioritization challenges affecting how a massive investment in tributary habitat restoration and RM&E can best be leveraged to help recover ESA-listed populations:

1. What are the tributary habitat *limiting factors* impacting the '**health**' of listed salmonids?
2. What tributary **improvement actions** would provide the most freshwater production benefits to listed salmonid populations?
3. How **effective** are tributary improvement actions at providing benefits to listed salmonids?

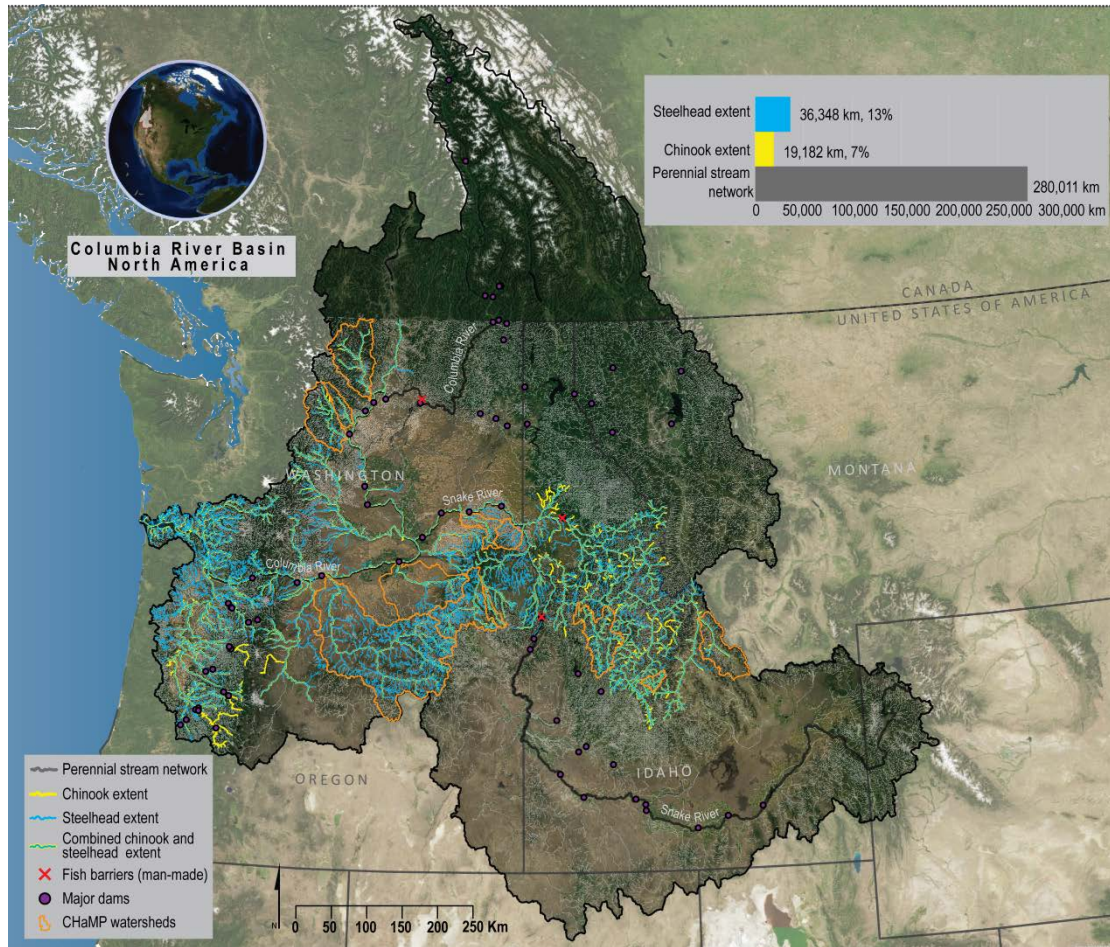


Figure 2 – The Columbia River Basin and US portion of its drainage network, showing in light grey the 280,011 km of perennial streams within the US, and what proportion is still accessible to anadromous steelhead (cyan) and chinook (yellow). Also shown are major dams, some of which (red x’s) are now barriers to their upstream migration. The 12 CHaMP study watersheds described in this paper are highlighted in Orange and correspond with key populations.

The first question considers the *condition* of tributary habitat and its impact on the *health* of listed salmonid populations (i.e. one informed by status and trend monitoring). The habitat condition part of that question is what the Bouwes *et al.* (2011) **Columbia Habitat Monitoring Program** (CHaMP – <http://champmonitoring.org>) was fundamentally designed to track and address. While monitoring data alone may give insights, additional analysis and interpretation of these data is required to inform assessments of habitat’s significance to populations, which is where the second question comes into play—how do we use the best available science and information to make planning decisions and design choices affecting

how restoration actions are implemented? Finally, the third question necessitates follow-up monitoring to determine whether or not investments in restoration are effective at improving the situation. Fundamentally, these management questions are asked at the ‘population-scale’, which typically corresponds to the extent of the drainage networks they occupy but does not resolve what individual fish are doing (simply sums up their abundances and survival between life stages across their extent). High-level policy may be concerned primarily with the bottom line population numbers that result in a viable, sustainable population target. However, those responsible for implementing specific tributary habitat restoration actions (i.e. KMQ 2) and those monitoring and evaluating the status and trends of these populations require insights across a range of scales to meaningfully address these key management questions.

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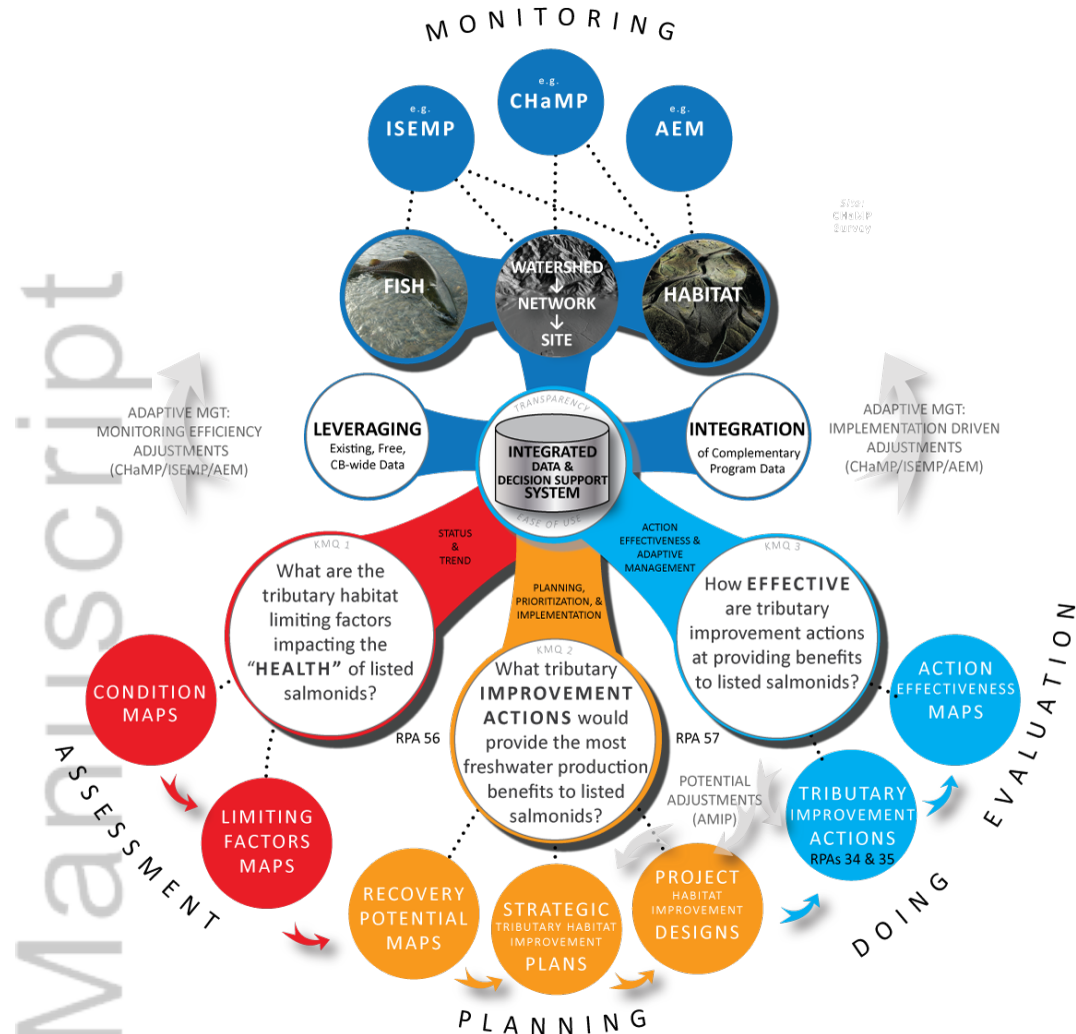


Figure 3 – The relationship between the key management questions in the CRB surrounding how tributary habitat conditions and restoration actions to improve them relate to the health of ESA salmonids. Monitoring programs like CHaMP and its affiliates (Integrated Status and Effectiveness Monitoring Program [ISEMP] and Action Effectiveness Monitoring [AEM]) provide data that when combined with other programs and existing data sets can be integrated and leveraged to address the key management questions in terms of assessment, planning and evaluation summary products (bottom circles).

### Ecohydraulic and Geomorphic Sampling and Analyses at Individual Sites

Contemporary ecohydraulic analysis takes place at the reach scale (see Table 1), and can resolve spatial patterns of habitat variability across individual raster cells, hydraulic units or

geomorphic units in two or three dimensions (Goodwin *et al.*, 2006; Pasternack, 2011). Within CHaMP, we collect field data at the reach scale, and in this section we describe what field data is collected, how we analyze it to produce detailed analyses at each sample site (i.e. reach), and which is frequently synthesized to summary metrics at the ‘Sites on Network’ scale (see Table 1).

### Habitat Sampling at Discrete Sites

A study design describes the overall strategy for *where* and *when* we take samples to ensure that robust statistical inference can be performed to tell us about a broader population of all possible sites (i.e. every reach segment on a drainage network). By contrast, the sample design defines *what* and *how* we perform the measurements at every sample site. If financial, human and time resources were infinite, we would use the same high-resolution sample design and census all 36,348 km of habitat everywhere. In some senses, remote sensing attempts to provide that census of habitat, but as we will explain - this is not pragmatic at larger (i.e.  $> 10^2$  km) network scales in bigger watersheds or across regional extents (i.e.  $> 10^3$  km<sup>2</sup>). However, even with all the current advances in remote sensing of rivers (e.g. Carbonneau *et al.*, 2011; Carbonneau and Piegay, 2012; Demarchi *et al.*, 2016b) no suite of sensors or platform is currently able to remotely sample the types of environments steelhead and Chinook use consistently, with either acceptable accuracy or adequate resolution and coverage to drive ecohydraulic analysis. As such, we are forced to strategically invest available resources in representative sampling methods that can inform questions of interest in a statistically robust way (i.e. in this example, those watersheds with ESA-listed populations of management concern).

### Study Design

Figure 2 shows the network extent of potential interest for Chinook and steelhead and identifies priority populations that correspond with the 12 CHaMP watersheds shown in orange. For most watersheds, the annual sampling effort is 25 sites (some have more). Thus, the study design needs to define *where* (i.e. which sites) that effort should be spent and *when* (i.e. how frequently)? The study design within CHaMP was developed using a GRTS (Generalized Random Tessellation Sample) design (Stevens and Olsen, 2004), which provides a spatially balanced random sample. If every site is sampled annually, we would only have 25 sites of spatial coverage per watershed. By introducing a temporal panel design

(Table 2) with both an annual panel of 15 sites surveyed each year and a three year rotating panel of 30 sites (10 per year, but resurveyed *only* every third year temporally), we increase the spatial coverage to 45 sites (Larsen *et al.*, 2007; Larsen *et al.*, 2008; Nahorniak *et al.*, 2015). The annual panel allows some insight into annual trends, while the rotating panel helps expand the spatial footprint of sites. Examples from two CHaMP watersheds are illustrated in Table 2 and Figure 4.

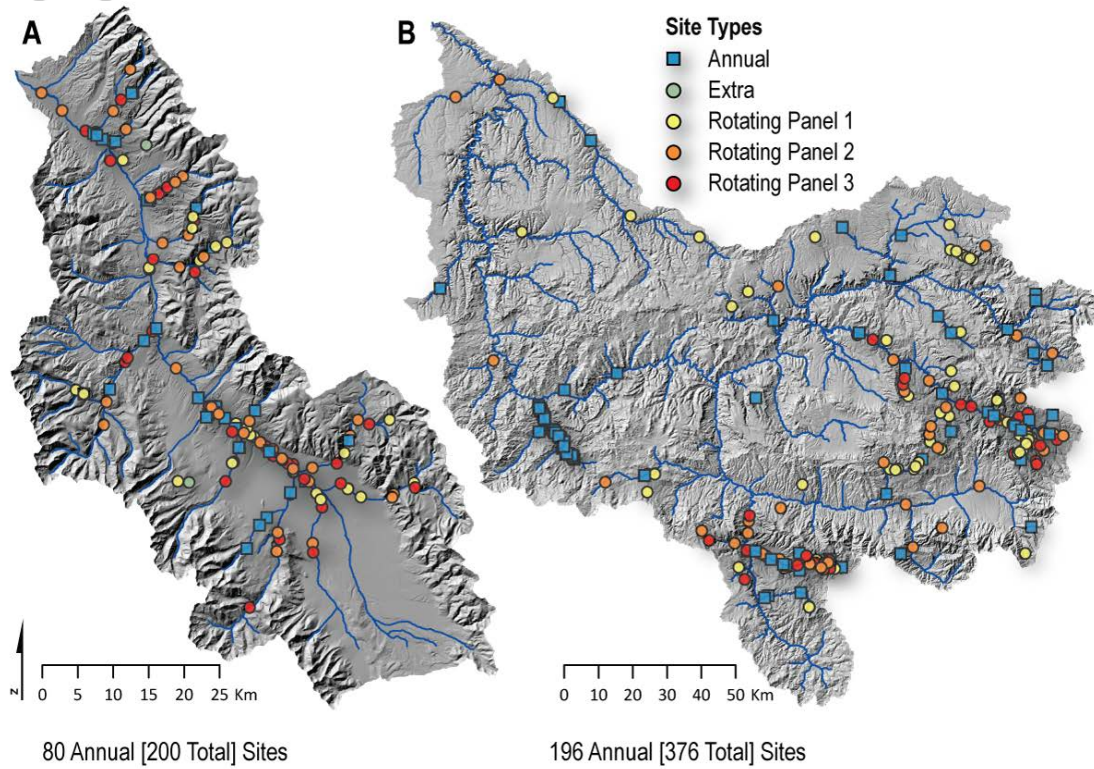


Figure 4 – Examples of GRTS study designs for two CHaMP watersheds (John Day in A and Lemhi in B). See Figure 2 for locations of watersheds and Table 2 for details of study designs.

Table 2 – Examples of three-year study designs over nine years contrasting 25 (top), 140 (middle – corresponding to Lemhi) and 156 (bottom – corresponding to John Day) sites of annual effort. See Figure 4 for corresponding spatial illustrations. Introducing rotating panels with less frequent resampling, helps expand the spatial coverage of sites.

Typical Survey Design	Year									Total Sites	
	1	2	3	4	5	6	7	8	9		
Annual	15	15	15	15	15	15	15	15	15	15	15
Rotating Panel 1	10	-	-	10	-	-	10	-	-	10	10
Rotating Panel 2	-	10	-	-	10	-	-	10	-	-	10
Rotating Panel 3	-	-	10	-	-	10	-	-	10	-	10
<b>Total Sites Surveyed</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>25</b>	<b>45</b>
Example Design 1 (e.g. Lemhi)	Year									Total Sites	
	1	2	3	4	5	6	7	8	9		
Annual	80	80	80	80	80	80	80	80	80	80	80
Rotating Panel 1	60	-	-	60	-	-	60	-	-	60	60
Rotating Panel 2	-	60	-	-	60	-	-	60	-	-	60
Rotating Panel 3	-	-	60	-	-	60	-	-	60	-	60
<b>Total Sites Surveyed</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>140</b>	<b>200</b>
Example Design 2 (e.g. John Day)	Year									Total Sites	
	1	2	3	4	5	6	7	8	9		
Annual	196	196	196	196	196	196	196	196	196	196	196
Rotating Panel 1	60	-	-	60	-	-	60	-	-	60	60
Rotating Panel 2	-	60	-	-	60	-	-	60	-	-	60
Rotating Panel 3	-	-	60	-	-	60	-	-	60	-	60
<b>Total Sites Surveyed</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>256</b>	<b>376</b>

### Sample Design

At each of the 905 sites that are surveyed in the CHaMP program, the Bouwes *et al.* (2011) protocol has been used to sample Wadeable reaches varying between 120 m and 600 m in length (roughly 20 bankfull widths). As of January, 2016, five years of monitoring using the study designs similar to those in Table 2 have been completed, translating to 2075 site visits, 833 revisits of 400 annual sites, and between one and two revisits to 585 rotating panel sites. The heart of the sample design is a continuous, topographic survey, which both paints a quantitative picture of physical habitat at the site and supports a multitude of subsequent ecohydraulic (e.g. Wheaton *et al.*, 2004) and geomorphic analyses (e.g. Wheaton *et al.*, 2010a; Wheaton *et al.*, 2010b) as well as over 150 River Bathymetry Toolkit (e.g. McKean *et al.*, 2009) derived metrics. In addition, a variety of ‘auxiliary’ measurements are made to collect data on other habitat variables commonly collected by traditional habitat sampling programs like the PIBO (Pacific-fish Inland-fish Biological Opinion; Heitke *et al.*, 2010) and

AREMP programs, which are not based around a topographic survey. The Bouwes *et al.* (2011) protocol and subsequent updates (e.g. CHaMP (Columbia Habitat Monitoring Program), 2013), which are routinely updated on <http://monitoringresources.org>, provide the full details of the sample design methods. Below, we provide brief overview of the topographic survey and habitat sample designs.

### ***Topographic Survey Methods***

Within each reach, a standardized, high resolution topographic survey is performed using a topographically stratified sampling scheme (Brasington *et al.*, 2000). A total station or rtkGPS is typically used as these are generally the only high-resolution survey techniques that will work across the full diversity of sites surveyed (Bangen *et al.*, 2014a). Each year prior to the field season, 15 to 20 survey crews undergo an intensive two-week training to ensure consistency in survey methods and approach (Bangen *et al.*, 2014b). At each reach, a local control network is established and maintained to ensure subsequent resurveys can be performed on the same control network. Real-world UTM coordinates are either tied into existing professional control networks, or more commonly approximately established using the CHaMP Transformation Tool (Wheaton *et al.*, 2012). Each survey point is attributed with feature codes to differentiate breaklines (e.g. top of bank, toe of bank), thalwegs, and other features that assist in producing more accurate DEMs (see §5 and Table 4, CHaMP (Columbia Habitat Monitoring Program), 2013). Crews are responsible for post-processing the surveys using the CHaMP Topo Toolbar (<http://champtools.northarrowresearch.com/>) as they are in the best position to ensure that the maps they produce are an accurate portrayal of what they measured in the field (Bangen *et al.*, 2014b). The primary outputs of this post processing are a 10 cm resolution digital elevation model (DEM) capturing detail within the bankfull channel(s) and adequate context on the floodplain for high-stage flood modelling and or geomorphic change detection of lateral erosion (Wheaton *et al.*, 2010b). After undergoing a rigorous series of quality assurance checks, the surveys are uploaded to cloud-based repository (<http://champmonitoring.org>) from which a series of morphometric and geomorphic change detection (GCD) analyses (using GCD software: <http://gcd.joewheaton.org>) are automated using cloud computing (e.g. EC2, using Amazon Web Services).

### ***Habitat Sampling Methods***

While the topographic survey is the most basic representation and mapping of physical habitat at a site, one member of the crew is focused on collecting a host of ‘auxiliary’ habitat data following the (CHaMP (Columbia Habitat Monitoring Program), 2013) protocol. This includes mapping channel units using Hawkins *et al.* (1993) and recording for each channel unit a mix of ocular estimates and measurements to quantify fish cover (Peck *et al.*, 2001), large woody debris (volumes and counts), and substrate composition. In addition, undercut banks are mapped and measured to assess fish cover (i.e. predation refugia). Pool-tail fines are quantified following Heitke *et al.* (2008) and pebble counts and embeddedness are measured in all riffles for understanding the adequacy of substrate to support spawning and egg survival, and flow refugia for juveniles in fast moving water. At the site level and extent, a field sketch site map is produced, and a series of photos are taken systematically. To characterize riparian conditions, in part for fish cover and terrestrial inputs and in part to explain temperatures, solar input is measured using a Solmetric Suneye™, and riparian structure is quantified using adaptations of the Peck *et al.* (2001) and (Kaufmann *et al.*, 1999) protocols. Onset Tidbit temperature loggers are deployed at every site following Isaak *et al.* (2010a) to produce continuous time series of water temperature to aide in bioenergetics and production modelling. Discharge is measured near the top of every site following Peck *et al.* (2001) to contextualize site visits and act as a boundary condition for hydraulic modelling. Conductivity and alkalinity are measured during the visit to provide crude water quality context. Finally, drift nets are deployed to measure the density of invertebrates drifting in the water column, a primary input to drift foraging models and a measure of food availability that informs model-based assessments of juvenile rearing capacity (Wall *et al.*, 2015).

### **Ecohydraulic Analyses at Discrete Sites**

After site visits are post processed, subjected to quality assurance checks, turned into DEMs and a variety of derivative products are produced (e.g. water depth maps, detrended DEMs, channel unit maps, etc.), ecohydraulic analyses are undertaken for each site. Our ecohydraulic analyses are dependent on multi-dimensional hydraulic model simulations (i.e. two-dimensional or three-dimensional computational fluid dynamic models) as inputs (Leclerc *et al.*, 1995; Benjankar *et al.*, 2015). When the computational meshes for these hydraulic models are discretized at high spatial resolution (i.e. 5 to 50 cm computational node spacing) and fed

with high quality topographic data (e.g. our 10 cm resolution DEMs), these models are capable of resolving hydraulic patterns at a scale that is relevant to fish (Wheaton *et al.*, 2010a).

One of the most time-consuming steps in hydraulic modeling is producing computational meshes. We have automated the process of computational mesh construction at all sites by using a simplified rectilinear (i.e. raster) meshing scheme, but using between 10 cm and 20 cm resolution node spacing. Such a high-resolution computational mesh was previously computationally costly, but with inexpensive cloud computing it is much more cost-effective than manual editing 1000's of coarser resolution meshes using unstructured and/or curvilinear meshing schemes. The meshing process is based on the CHaMP survey formats, and set boundary conditions from auxiliary measurements like roughness, discharge entering the reach, and corresponding water surface elevation at the downstream end of the reach. Moreover, because crews topographically survey the water's edge and major breaks in water surface topography, water depth maps are derived by subtracting the DEM from a water surface raster, and these are directly compared to hydraulic models for depth validation. The boundary conditions (discharge, downstream water surface elevation), and resulting hydraulic model solution, reflect low-flow conditions due to CHaMP's survey timing (i.e., summer/fall). However, if real or synthetic rating curves (i.e. discharge vs. stage) are available for sites, the hydraulic model can be run at a range of flows, for instance in support of assessments quantifying varying habitat availability at different flows. Leveraging the open-source Delft3D hydraulic model code (Deltares, 2010), we have now fully automated a cloud computing workflow. To date, we have used this workflow to automatically generate models and run tens of thousands of simulations at over 2075 visits at 985 sites. The quality of ecohydraulic model simulations is highly dependent on hydraulic model accuracy (Pasternack *et al.*, 2006a; Shen and Diplas, 2008), which for these systems is most influenced by the quality of the topographic boundary conditions and the representation of structural elements impacting the flow field in the computational mesh. Hydraulic model validation is performed by independently measuring depth fields (done with all CHaMP surveys), spot and cross sectional measurements of velocity, and mapping of large flow separation points and shear zones.

For the purposes of this paper and illustration, we only present and describe summer and fall baseflow simulations in these tributary habitats with unregulated flow regimes, which are representative of habitat conditions at the time of these surveys. The hydraulic models from CHaMP surveys can be reliably run at flows up to a ‘bankfull’ discharge contained within the channel, and at some sites (depending on quality and extent of floodplain survey) limited overbank flows can be resolved. Delft3D and similar hydraulic models based on Navier-Stokes approximate solutions to the equations of motion, generally perform reasonably well when the quality of the topographic data is high. DEM quality is quantified by building DEM error models (output of  $\pm$  m) using fuzzy inference systems (FIS) following Bangen *et al.* (2016) and a method developed by Wheaton *et al.* (2010b). Currently, DEM error is primarily used to inform geomorphic change detection, but its significance can be explored (e.g. Legleiter *et al.*, 2011) and is generally not limiting for these types of surveys (Pasternack *et al.*, 2006b).

#### Spawning Life stage – HSI and FIS modelling

Salmonid spawning typically takes place at locations (i.e. on riffles and pool-exit slopes) where multi-dimensional hydraulic models can perform reasonably well (Pasternack *et al.*, 2006b) if driven with accurate topographic, roughness and flow boundary conditions. In CHaMP, we have automated the simulation of two common approaches to modeling spawning habitat quality (Ahmadi-Nedushan *et al.*, 2006; Pasternack, 2011) – the habitat suitability index (HSI) method (Raleigh *et al.*, 1986) and a fuzzy inference systems (FIS) method (Ahmadi-Nedushan *et al.*, 2008). Both approaches are driven on the physical side by substrate character (a CHaMP ‘auxiliary’ variable resolved at the geomorphic unit level) and the hydraulic model outputs of depth and velocity. On the biotic side, multiple physical variables (typically depth, velocity and substrate) are related to habitat quality with slightly different approaches that both score habitat quality on a 0 to 1 scale (1 being highest quality). In CHaMP applications, the models are applied at every ‘wet’ computational node.

For the HSI method, the overall ‘suitability’ or quality index for a node is determined by either averaging or multiplying individual scores of habitat quality for each physical variable (e.g Figure 5A-C) obtained from a univariate habitat suitability curve (HSC). The HSI/HSC



framework is one of the earliest manifestations of an ecohydraulics modelling approach, and was popularized via the instream flow incremental methodology (IFIM) and physical habitat simulation (PHABSIM) system during the 1980s and 1990s (Reiser *et al.*, 1989). While there is some healthy skepticism of HSC-based approaches (e.g. Railsback, 2016), many investigators have shown these models can provide robust predictions when driven by accurate, multi-dimensional hydraulic models at ecologically meaningful scales (e.g. Leclerc *et al.*, 1995; Crowder and Diplas, 2000; Shen and Diplas, 2008) and with HSCs derived in or appropriate to the system of study.

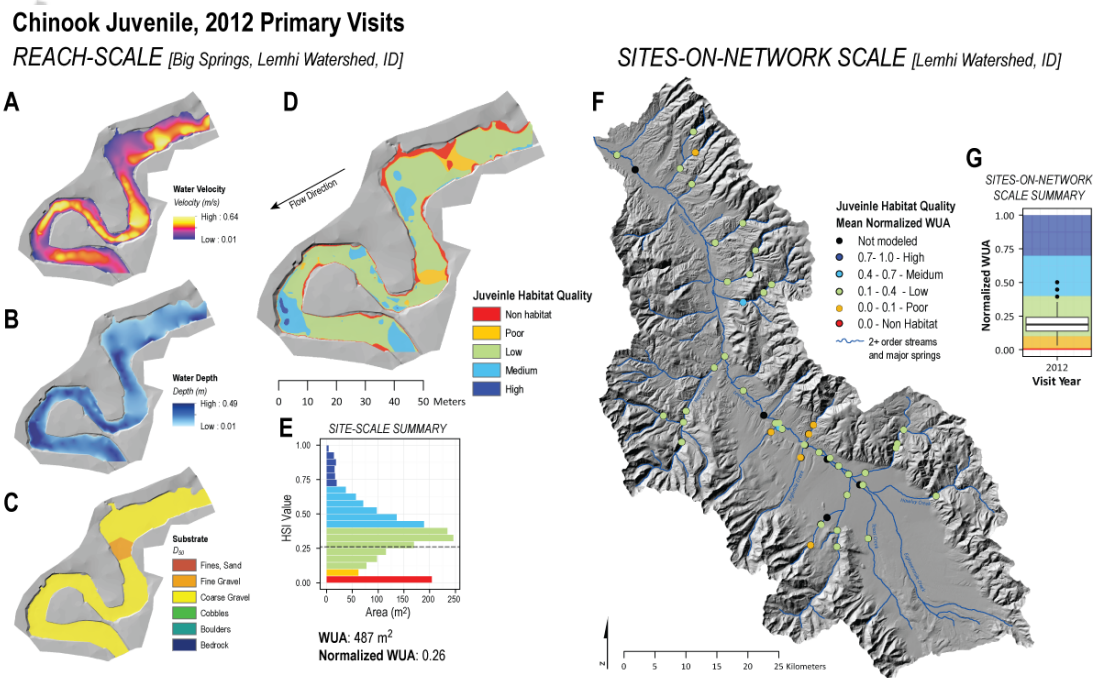


Figure 5 – Illustration of a site-scale juvenile rearing habitat HSI simulation (D), but driven by site-scale inputs of velocity (A), depth (B) and substrate type (C) that drive HSI or FIS simulations. Site-summary-scale syntheses of ecohydraulic model outputs (E) help distill the rich information down to summary metrics (in this case normalized weighted useable area) to something that can be displayed for all CHaMP sites (F) at the sites-on-network-scale. Using distributions of site scale summaries, box plots for WUA predictions at all surveyed sites can be made (e.g. G).

For the FIS method, relationships between habitat variables and suitability are represented in an inference system or rule table. The rules (or rows) in the table map out how different

combinations of inputs, represented as linguistic variables (e.g. velocity could be ‘low’, ‘medium’ and ‘high’ or ‘slow’, ‘average’ and ‘swift’), result in a given habitat quality (e.g. ‘swift velocity’, ‘shallow depth’ and gravel substrate equates to ‘high quality spawning habitat’). The implementation of the inference system is made ‘fuzzy’ by allowing overlapping membership for different combinations of variables, with the degree of overlap capturing uncertainty in input-to-output mapping. For any combination of inputs for which an input has ‘overlapping’ membership, multiple rules will apply, allowing for a more nuanced representation in the continuous output of the model, even when ‘defuzzified’ back to a single crisp value. Additionally, when developing FIS models, analysts/experts must inherently consider the multivariate nature of habitat inputs—the same way in which fish experience conditions in nature—whereas HSC are typically derived/applied on a univariate basis.

We implement both styles of models in an open-source, database-backed, geospatial habitat model interface we call the Fish Habitat Model (FHM: <http://fhn.northarrowresearch.com>). The FHM has an easy-to-use GUI interface that individual simulations and scenarios can be run in, but because multiple steady-state model simulations and numerous models are run for every site visit, the database backbone of the FHM is tailored for automating and batching multiple simulations simultaneously. FHM is implemented by CHaMP using cloud-computing via Amazon Web Services (AWS), and this allows us to quickly produce site-scale ecohydraulic simulations at every site (e.g. Figure 5D) at all CHaMP sites in a basin (e.g. Figure 5F shows site-summary scale results for all 2012 sample sites from Figure 4A).

#### Juvenile Rearing Life Stage – NREI modelling

Juvenile rearing is a critical time during the anadromous life cycle. Young fish are trying to eat and grow sufficiently to improve their chances of successfully navigating the downstream journey to the ocean, all while avoiding becoming prey themselves (Dill and Fraser, 1984). Both HSI (e.g. Raleigh *et al.*, 1986) and FIS models (e.g. Atlantic salmon, (Mocq *et al.*, 2013)) exist for juvenile salmonid rearing, and we use virtually any of these within CHaMP, applying them via the FHM model framework described above (e.g. Figure 5). In contrast to spawners, for which habitat suitability can be reasonably inferred based on few physical variables (Gallagher and Gard, 1999), suitability for juvenile salmonid rearing is governed by a more complex set of factors (Rosenfeld *et al.*, 2005). HSI/FIS models tend to provide a

simplistic view of conditions, representing by proxy the availability of depth/velocity combinations commonly occupied by fish. However, because most HSI/FIS approaches do not include important determinants of habitat use like food availability or temperature as predictors, the quality/capacity of habitats for juveniles can be better approximated using a more mechanistic approach (Railsback, 2016). For example, an approach that explicitly considers the bioenergetics of this life stage by accounting for the role stream temperature plays in metabolism, the availability of food (i.e., drifting macroinvertebrate prey), the probability of prey capture, and the costs of locomotion in a stream environment (i.e., swimming costs) (Hughes *et al.*, 2003; Wall *et al.*, 2015).

In CHaMP, we have operationalized approaches inspired by (Hayes *et al.*, 2007b) and elaborated in (Wall *et al.*, 2015). These approaches are known as NREI – Net Rate of Energy Intake – models, because they calculate the net energy balance from an individual fish’s perspective if it were to maintain position within every computational node of the wetted channel. Briefly, given the local depths and velocities from hydraulic model simulations, the size of fish under consideration, and water temperature, the NREI model calculates the cost of swimming or holding position in a given locality. Then, using CHaMP data on invertebrate drift density (individuals / volume), the modeled fish’s swimming ability, and also its reactive distance to prey items, the model provides an estimate of the number of prey items that a fish can access at different nodes per unit time. Given a probability of capture for prey encounters, a likely energy intake can be estimated. Finally, by comparing the cost of swimming (in Joules), to the energy gain from capturing drifting prey (also in Joules), a net rate of energy intake can be calculated and zones that are energetically profitable, deficient, and neutral emerge (Figure 6C -> Showing an analogous output at the same site as Figure 5D).

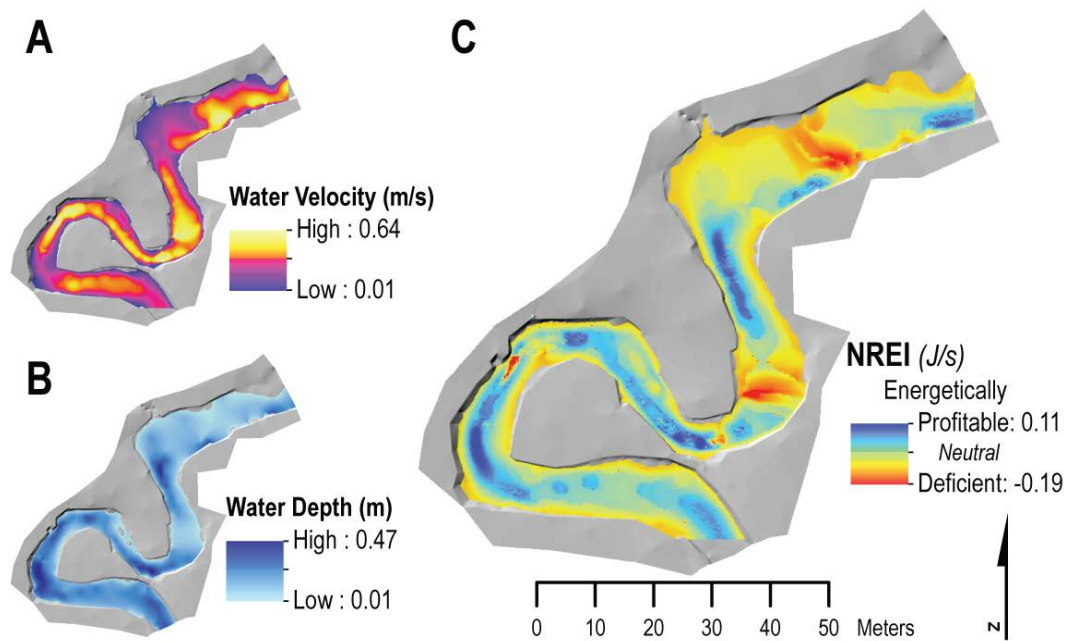


Figure 6 – Example of site-scale NREI output (c), based (in part) on 2D hydraulic model predictions of velocity (A) and depth (B).

Ecohydraulic models can be used to describe microhabitat scale (or hydraulic unit as defined in Table 1) habitat utilization by salmonids at a higher resolution than can be afforded by geomorphic unit or reach scale summaries. They can provide mechanistic insight into measures of habitat complexity that remain elusive at these coarser scales. For example, NREI models would generally predict that salmonids will minimize energy expenditure by holding in low velocity positions that offer access to high velocity zones within their reactive distance to maximize prey encounters (Hughes, 1998; Hughes *et al.*, 2003). These shear zones (cf. Wheaton *et al.*, 2015) are often found along lateral transition of geomorphic units, such as between bar forced pools and structurally forced bars. Simple reach level metrics such as pool frequency or pool area, commonly used in traditional empirical models describing fish-habitat relationships may not be adequate to define these more descriptive measures of habitat quality. For example, we used a simple path analysis to describe NREI estimates of carrying capacity at 64 CHaMP sites. Path analyses regress explanatory variables not only to the predictor variable but other explanatory variables, allowing for indirect pathways. We created a variable to describe shear zones, which was based on a roving window summary across the velocity raster produced by hydraulic model. The window size equaled the reactive distance

of a juvenile steelhead. Gradients higher than 0.2 m/s were deemed shear zones, the sum of which produced the total area of shear zones in a reach. Gradient, shear zones, and pool frequency were used as explanatory variables. Gradient predicted pool frequency and shear zones. Pool frequency helped explain shear zones as well but did not explain carrying capacity. Only shear zones predicted capacity, suggesting that after accounting for shear zones, pools themselves are not the strong predictor of salmonid habitat quality as commonly assumed. Because the arrangement of geomorphic units can describe general hydraulic patterns (Moir and Pasternack, 2008; Wyrick and Pasternack, 2014; Wyrick *et al.*, 2014), a more direct metric derived through models like NREI, such as area of transitions zones or the perimeter of influential geomorphic units, can be developed. We are currently assessing how geomorphic unit assemblages can be used to approximate foraging efficiency.

### Site Capacity Estimates

To address the key management questions outlined above, we need to translate these quantitative habitat measurements into estimates of capacity – or how many fish these reaches can support for adult spawning and juvenile summer rearing life stages. Ecohydraulic models are not typically used in this way, tending instead to focus either on indices of habitat quality (e.g. Hanrahan *et al.*, 2004) or a bioenergetics assessment (Hayes *et al.*, 2007a). However, if assumptions are made about the size of habitat a fish of a given size occupies (i.e. a territory rules) when drift-feeding, holding position, or spawning, HSI/FIS and NREI models can be used to estimate of the maximum number of fish that a site can support (i.e. capacity (Hanrahan *et al.*, 2004; Hayes *et al.*, 2007b)). In contrast to purely geometric methods (i.e., capacity = area/territory size), which are likely to overestimate capacity, habitats that are energetically unfavorable (i.e., NREI juvenile capacity) can be excluded. For habitats that are ‘suitable’ or bioenergetically favorable, the relative contribution of each cell to total capacity can be simply scaled or weighted in a manner commensurate with its habitat quality (i.e., HSI/FIS spawner capacity). An example of this capacity estimation approach is illustrated in Figure 7 for NREI-based predictions, based on an approach implemented in Hayes *et al.* (2016) and Wall *et al.* (2015). The output of such an approach is a predicted upper number of fish the site can support (i.e. capacity), and to make these capacity estimates inter-comparable

across sites they can be normalized into a lineal density (e.g. fish/m in Figure 7B) by dividing by site length, or into an areal density (i.e. fish/m<sup>2</sup>) by dividing by wetted area.

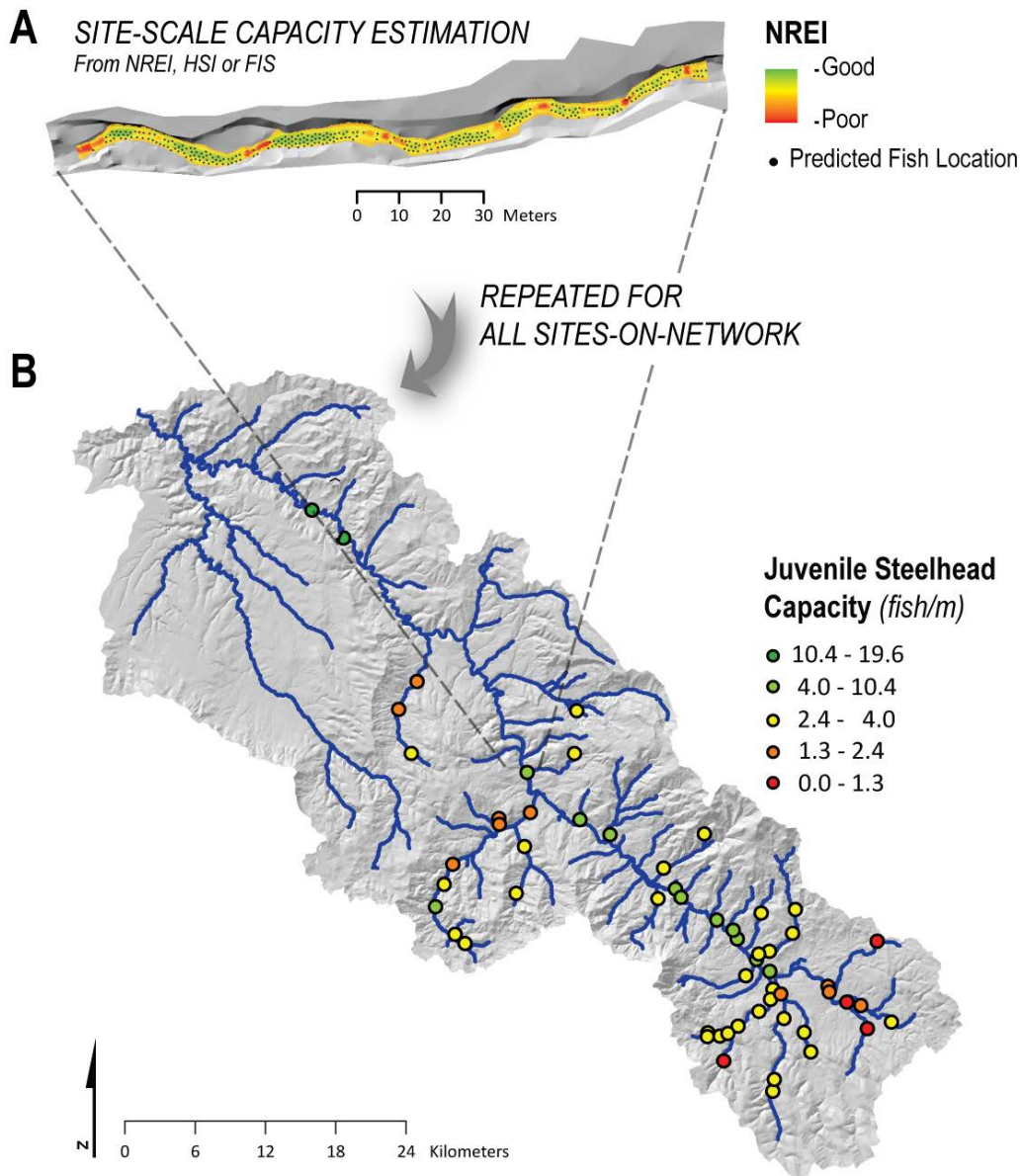


Figure 7 – Example of estimation of capacity at single site (A) and how summing the values, dividing by reach length, capacity can be expressed as a fish density at all sites-on-network scale (B).

### Upscaling Site-Scale Information & Network Modeling

While site-scale (as defined in Table 1) ecohydraulic model results provide useful insights and can inform specific restoration designs or effectiveness monitoring at those sites, their

extent is insufficient for answering key management questions across a population's full range in a drainage network (Isaak *et al.*, 2016). Synthesis of site-scale analyses into descriptive site-summary-scale metrics, like capacity, are inadequate to assess an entire population. Even for study designs like used in CHaMP, where site-summary-scale metrics exist at a number of sites across a network (i.e. sites-on-network-scale), some upscaling methods are still required to inform population level questions. First we describe three network co-variates that could be used to support juvenile rearing capacity estimation, and then we contrast the different upscaling techniques that use these. We discuss three approaches whereby multiple sites on the network are used to inform population level questions: design-based and correlative and causal model-based estimation. All three approaches require network-scale co-variates.

### Network Co-Variates

While there are many potential network scale co-variates that could be used, we highlight three key network co-variates, an example of which is illustrated in Figure 8A-C.



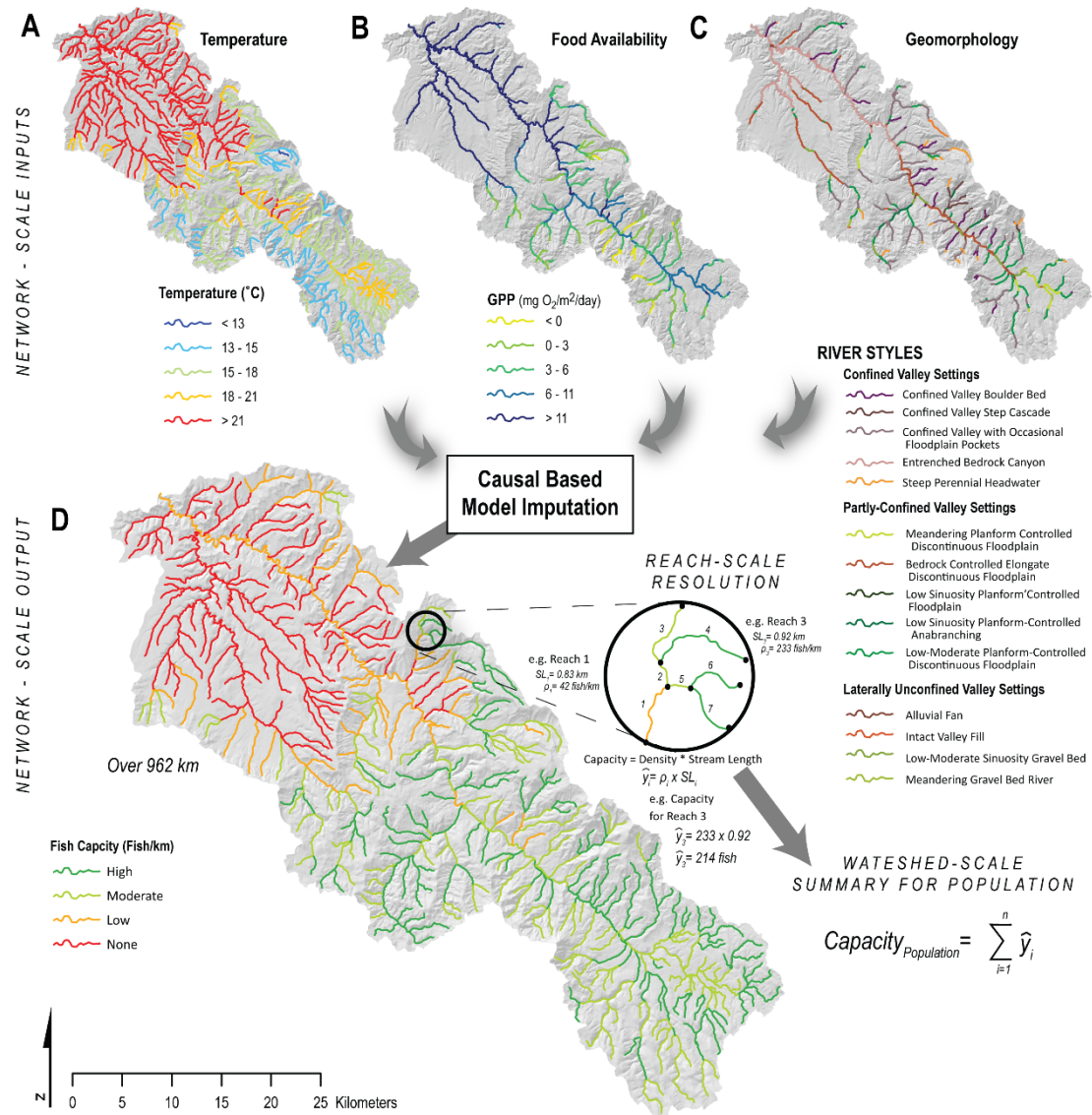


Figure 8 - Illustrative example of network-scale inputs of temperature (a), gross primary production (b; i.e. proxy for food), and river styles (c) used to inform a model imputation to drive a network-scale fish response output (d).

### Physical Template - Geomorphic Reach Type & Habitat Condition

The local hydraulics, channel morphology, substrate type, and structural elements (e.g. wood) define the physical habitat setting perceived by salmonids. Site-scale ecohydraulic model outputs largely reflect these variations between hydraulic units, which form the building blocks of geomorphic units. Geomorphic units are the building blocks of a reach (Fryirs and



Brierley, 2013), but also the nexus of a reach's identity (Montgomery and Buffington, 1998) and the microhabitat considerations of salmonids. The identification of geomorphic unit assemblages within distinctive reach types throughout the stream network could provide extremely important information on the distribution of salmonid habitat (Hankin and Reeves, 1988; Schwartz, 2016). Censusing geomorphic units is possible at the reach-scale or site-scale (Wyrick *et al.*, 2014), and long reaches can be sampled with rapid assessments (Camp and Wheaton, 2014; Bouwes *et al.*, 2016). For example, Blanchard (2015) censused over 30.5 kilometers of streams using the Wheaton *et al.* (2015) taxonomy for geomorphic units, representing 197 CHaMP sites, but taking the equivalent effort of sampling 30 sites using the CHaMP protocol. However, censusing an entire network at a population scale (e.g. McMillan *et al.*, 2013) is typically impractical. As such, we need a means to predict what local hydraulics, geomorphic units and structural elements will be present, but for every reach in a network (e.g. the 962 km in Figure 8 or the 36,000+ km in Figure 2). Numerous potential reach-typing classification schemes exist see Buffington and Montgomery (2013). There are various network models for predicting grain size throughout a network (e.g. Snyder *et al.*, 2013), and even suitability for spawning (Buffington *et al.*, 2004). However, of those reach-typing schemes that exist, only a few can be applied across a network (e.g. Beechie and Imaki, 2014) and most don't necessarily predict the expected geomorphic units and structural elements (Kasprak *et al.*, 2016).

We use the Brierley and Fryirs (2005) River Styles framework to model reach types, geomorphic condition, and recovery potential (e.g. O'Brien and Wheaton, 2015; O'Brien *et al.*, In Review). Similar, promising frameworks could be potentially substituted for at least the reach-typing components (e.g. Bizzi and Lerner, 2012; Bizzi *et al.*, 2013; Beechie and Imaki, 2014; Gurnell *et al.*, 2015; Demarchi *et al.*, 2016b; Demarchi *et al.*, 2016a; Kasprak *et al.*, 2016), but here condition assessment and recovery potential are critical to addressing key management questions. Readily available remotely-sensed data such as 10m DEMs, geology and LANDSAT-derived vegetation layers, satellite and aerial imagery (e.g. Google Earth), along with overflights and on-the-ground validation can all be leveraged to produce network scale geomorphic assessments (e.g. Figure 8C). Geomorphic condition (cf. Fryirs, 2015) combined with reach type is likely to be the best network-scale predictor of fish habitat character in individual reaches. Blanchard (2015) found that reach type alone was an adequate

discriminator of fish habitat, but this was likely because most reach types were only present in one condition variant in the study watershed. The combination of reach type and condition form a prediction of what fish habitat looks like on a reach-by-reach basis at the network-scale that can be readily validated and spot-checked with site-scale CHaMP surveys or reach-scale rapid assessments (Figure 9). By contrast, an honest appraisal of recovery potential (Figure 9) can form the basis for development of a range of realistic alternative restoration scenarios, which can then be used to explore potential capacity gains and subsequent fish population responses from various management alternatives. Too often, restoration planning is based on the wishful thinking that a historic condition is an attainable target (Roni and Beechie, 2013), whereas with the Brierley and Fryirs (2005) assessment of recovery potential, the contemporary boundary conditions help determine whether a not a historic condition or some intermediate condition is possible.

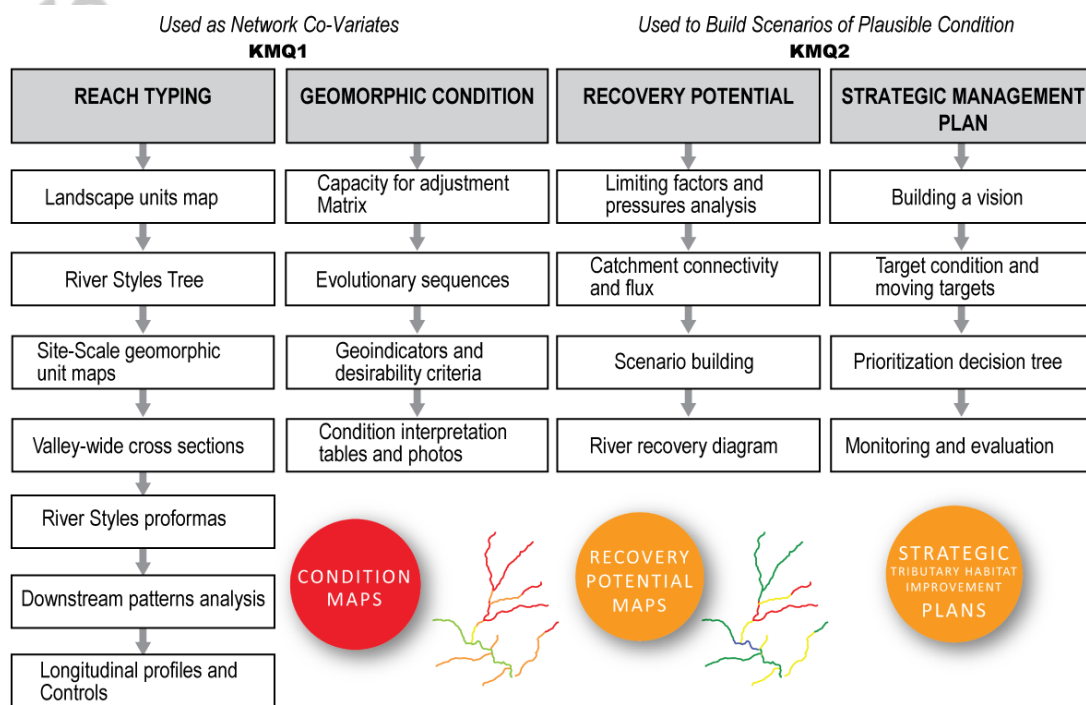


Figure 9 – Illustration of four key outputs of geomorphic analysis (grey boxes), and supporting analyses (white boxes) used to produce network scale reach-type and condition maps, which form a combined co-variate for upscaling. See Figure 3 for how these outputs support addressing key management questions (KMQ) 1 and 2. Figure adapted from O'Brien *et al.* (In Review).

## Temperature Modelling

Stream temperature is an extremely important factor necessary in the description of salmonid habitat. Temperature affects the physiological processes of all organisms, and thus stream temperatures affect stream food web dynamics, life history strategies, and behavior (Poole and Berman, 2001). Temperature directly influences fish survival, and consumption and growth rates (Elliott, 1976; Kitchell *et al.*, 1977). With the onset of inexpensive temperature loggers, stream temperature monitoring is ubiquitous (Isaak *et al.*, 2010a). Large datasets have led to a better understanding of the factors controlling stream temperatures, and the development of models to predict temperatures across the stream network (Gardner *et al.*, 2003; Webb *et al.*, 2008; Isaak *et al.*, 2010b; Pike *et al.*, 2013). Some models are highly mechanistic and can evaluate the expected impacts of alternative restoration scenarios (Boyd and Kasper, 2003), however such modeling is extremely data and labor intensive to implement over large extents. Remote sensing of water and land thermal signatures have also been leveraged to extrapolate instream temperature logger data to stream networks (McNyset *et al.*, 2015; Vatland *et al.*, 2015). Here, we use the temperature modeling approach of McNyset *et al.* (2015) that we use in estimating stream temperature across the network.

McNyset *et al.* (2015) has developed a stream network model that estimates daily temperatures over the entire year (e.g. Figure 8A). Remotely-sensed Land Surface Temperature (LST) is collected from the U.S. National Aeronautics and Space Administration's (NASA) Moderate Resolution Imaging Spectroradiometer (MODIS) satellites. Daily LST data is obtained at a resolution of 1 km over most of the planet. McNyset *et al.* (2015) found daily and 8 day averaged LST correlated highly with these same metrics from stream temperature loggers. Regression models were used to estimate temperatures throughout the network, with cross validation indicating these relationships were accurate and robust with as few as four stream temperature loggers. Some of the factors influence LST are air temperature, climate, vegetation, surface geology, elevation, and physiography, which are also several of the controlling factors of stream temperature thus explaining the strong correlations observed. While this model is capable of predicting stream temperature throughout the watershed, addressing KMQ1, the ability to predict changes due to restoration are not possible. We have used the Heat Source model of Boyd and Kasper

(2003) to predict stream temperature across the smaller extents where restoration actions are implemented when addressing KMQ2.

#### Primary Production – Food Proxy

Although stream food webs are inherently complex, integrating basal energy fluxes from multiple habitats (e.g., riparian vegetation or detritus, terrestrial invertebrate resources, and allochthonous production; Wallace *et al.*, 1997; Baxter *et al.*, 2005), the total production of carbon by aquatic primary producers (gross primary production, GPP) provides a good index of the potential for streams to produce invertebrate prey resources for fish (Cummins and Klug, 1979; McCutchan and Lewis, 2002) as it is a measure of the abundance of basal food web resources in streams frequently used by rearing steelhead and salmon (e.g. midsize channels with partially open canopies). Thus, modeling basal food web resources (i.e. GPP) can provide a reasonable proxy for fish food, and has been shown by Saunders *et al.* (in Review) to correlate well with fish abundance. Further, GPP is responsive to changes in land use (Bernot *et al.*, 2010; Griffiths *et al.*, 2013), stream and riparian restoration (Adams *et al.*, 2002; Riley and Dodds, 2012; Giling *et al.*, 2013; Roley *et al.*, 2014), and geomorphology (Coleman and Dahm, 1990), and thus can account for factors known to influence spatial variation in fish populations. Primary production can be approximated from dissolved oxygen time series (Cox, 2003; Grace *et al.*, 2015), as dissolved oxygen is a proxy for whole system metabolism (the combined processes of primary production and ecosystem respiration; Odum, 1956; Ortiz-Zayas *et al.*, 2005). Although the dynamics of stream and riparian ecosystems that influence primary production are complex, the primary factors driving large-scale spatial variation (e.g. solar exposure, nutrient concentration, and water temperature, see Mulholland *et al.*, 2001; Bernot *et al.*, 2010) can be assessed remotely (nutrients [conductivity/alkalinity] (e.g. Olson and Hawkins, 2012), sunlight (Fu and Rich, 2002), or mapped to stream networks temperature (e.g. McNyset *et al.*, 2015) to continuously predict GPP throughout river networks. Thus, we make continuous predictions of primary production throughout a river network using predictor variables that make conceptual sense to model spatial variation in potential food resources for fish populations (e.g. Figure 8B).

## Design-Based Estimation

In order to provide population estimates of the mean and variance of any metric of interest at the watershed-scale from site-level surveys, a representative probabilistic sample design is required. Uniform and non-uniform (e.g. stratified) probabilistic sample designs are commonly used to distribute sites throughout a watershed (Nahorniak *et al.*, 2015). In most cases, CHaMP uses a stratified, spatially balanced, probabilistic sampling design. Weights based on the inclusion probabilities are used to provide unbiased weighted estimates of the mean and variance of site level metrics when summarized across the watershed. This approach has been shown to be an efficient and precise means to describe the status and trends of site-level salmonid habitat metrics at the watershed extent with 30-50 sites sampled (Larsen *et al.*, 2004). Whether the sampling design captures the variability across a watershed for more multivariate metrics such as carrying capacity estimates from ecohydraulic models requires further investigation.

While a design-based estimation approach using capacity may provide summary information on status and trends of populations (KMQ1 and potentially KMQ3), the watershed-scale resolution of this upscaling does not resolve what is happening at individual reaches that a population may utilize (i.e. network-scale) to address where restoration should be implemented and what impairments restoration should target (KMQ2). Given that restoration priorities are planned and implemented at specific locations on the ground, we assert that there is a pressing need for reliable information at the network-scale (i.e. watershed extent, but reach resolution) about where capacity is limiting and if and how it may be expanded (Benda *et al.*, 2007; Macfarlane *et al.*, 2015; Macfarlane *et al.*, In Review). We call this the ‘dots to network’ problem – where dots are sites.

## Imputation Techniques

All techniques for tackling the ‘dots to network’ problem are basically spatial modeling exercises. In statistics, the act of substituting a reasonable estimate for missing values is known as imputation (Li and Parker, 2008). Here, we wish to impute values of capacity spatially across a network, using some additional information at broader spatial extents from remote sensing, while properly accounting for sample design (Nahorniak *et al.*, 2015). Spatial models that exploit spatial autocorrelation may support imputation in the form of

kriging (e.g. Ver Hoef *et al.*, 2014). However, kriging is not a feasible approach when spatial autocorrelation can neither be assessed nor exploited because points are measured at too coarse of a resolution, or data are obtained from spatially balanced samples, as in CHaMP. When attempting to model spatial patterns continuously (either in x-y space as in rasters, or longitudinally along a line in network space), imputation can be done with the assistance of some informative covariates that correlate to capacity estimates (e.g. Figure 8 - Illustrative example of network-scale inputs of temperature (a), gross primary production (b; i.e. proxy for food), and river styles (c) used to inform a model imputation to drive a network-scale fish response output (d).). This assisted form of extrapolation is the basis for spatial or geographical imputation (Henry and Boscoe, 2008).

The choice of covariates for any imputation exercise will necessarily depend on the variable that is being imputed. A network covariate should be a variable that can be measured or modeled where site samples do not exist. Furthermore, the covariate should have information content such that there exists some non-zero correlation with the metric to be imputed. Correlative approaches test multiple potential covariates to find correlates to the variable to be imputed. Causal approaches base covariate selection on a process-based understanding of how they relate to the variable of interest. Final covariates will be empirically trained, calibrated, and/or simply validated to the imputed variable.

#### Correlative Model-Based Imputation

Empirical models can be derived that relate ecohydraulic estimates to globally available attributes, and then use these empirical models to estimate CHaMP metrics at unmeasured reaches. For simple linear models, we use model assisted regression to properly account for sampling design in the construction of empirical models. For more complex modeling techniques, Nahorniak *et al.* (2015) have developed a methodology called inverse probability boot-strapping (IPB) to properly account for sampling design while using model based statistical techniques. An ideal empirical model is unbiased across spatial scales of interest, and relationships observed within the data set over which the model is fit must be consistent at any spatial scale where the model is to be applied. Careful analysis of residuals must be performed, and in some cases models must be optimized to the spatial scale over which they are to be used for prediction.

As an illustrative example here, we developed a model-assisted multiple linear regression to predict HSI derived steelhead spawner wetted usable area (WUA/m; similar to Figure 5) from network-scale covariates describing attributes such as valley class, human disturbance, natural landform classifications, elevation, drainage area, slope, stream flow, stream width, etc. Variable selection was performed using an iterative procedure to minimize model AIC. Weights for the model-assisted regressions were proportional to the inverse of the site-level sample inclusion probabilities. Cross-validation between HSI model estimates of WUA and the correlative model predictions are somewhat in agreement across several watersheds in the Columbia River Basin (Figure 10). Because the model was developed over a wide range of conditions, imputation to locations not surveyed should be more reliable. While this approach holds promise in providing spatially-explicit estimates of carrying capacity at the network-scale, caution must be exercised, especially when extrapolating this model in localities it was not empirically developed for or for considering alternative restoration scenarios.

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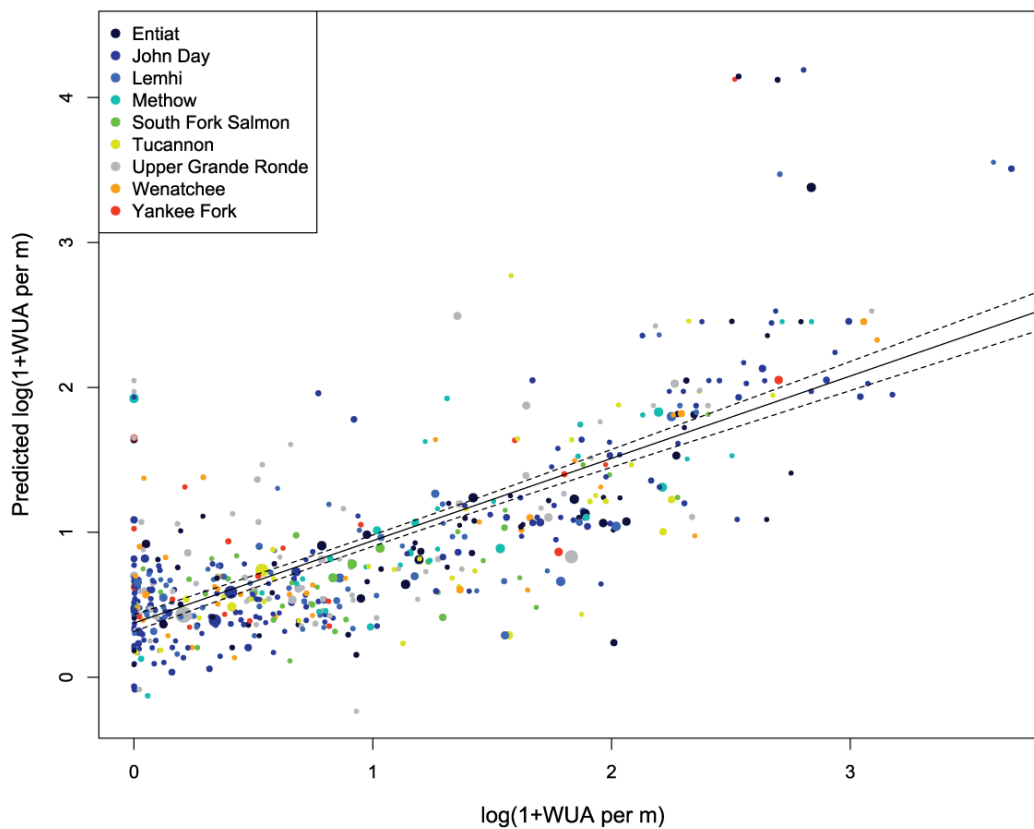


Figure 10 - Site-Summary-Scale measured versus regression model predicted  $\log(1+ \text{weighted usable area (WUA) per meter of stream length})$ . Dot size is proportional to sample weights for the model assisted regression.  $R\text{-squared} = 0.61$ .

#### Causal Model-Based Imputation

To identify habitat impairments and plan and test for appropriate restoration strategies to benefit populations, we posit that estimates of carrying capacity at the network-scale will be most informative. While correlative approaches, like above, may have the ability to create continuous estimates of carrying capacity of acceptable accuracy (e.g. Figure 10), the ability to manipulate covariate values to reflect potential changes due to restoration may not be possible (Brierley and Fryirs, 2005), or their prediction may not be defensible. To protect against spurious correlations, increase accuracy, and target variables subject to restoration, a process-based understanding, stemming from a strong and defensible theoretical foundation, of how the network scale variables influence carrying capacity is recommended.



Ecohydraulic site-scale models may help guide the development of network-scale models that inform the relevant reach scale metrics defining fish habitat.

For example, the general inputs for a site-scale NREI can be summarized into inputs of food, temperature, and channel morphology and substrate as they pertain to hydraulics. The goal with a causal model approach would be to create network extent of these metrics resolved at the reach scale (Figure 8A through C). At CHaMP sites, temperature is collected from temperature loggers and summarized (e.g. mean daily temperature). McNyset *et al.* (2015) demonstrated that MODUS satellite information of ground temperature taken daily is highly correlated with data logger temperature at CHaMP sites, and that this information can be extrapolated across the network with a high accuracy. Similarly, gross primary production estimated from DO sondes (see § Primary Production – Food Proxy above) could be predicted by site-level drivers like light, nutrients, and temperature, or remotely sensed/globally available proxies of these same variables (Šímová and Storch, 2016). Indeed, recent work using data from CHaMP sites suggests that causal models built on field measurements or globally available analogs (network-level GIS layers) predict GPP with similar accuracy and precision (Saunders *et al.*, Unpublished Manuscript), opening a door to network-wide prediction of energy availability. Finally, our investigations with ecohydraulic models at CHaMP sites suggest that the hydraulic patterns such as shear zones, found near geomorphic unit transitions, may be more important than geomorphic unit themselves (e.g. pools per kilometer) in determining habitat quality. We are currently refining the relationship between hydraulic patterns, NREI values, and geomorphic unit assemblages to capture this complexity rather than relying simply on metrics such as pool frequency or pool area. As discussed above, the River Styles framework delineates reach types with predictable geomorphic unit assemblages throughout the stream network. CHaMP surveys provide geomorphic unit assemblages that can be derived through topography and are consistent with the River Styles framework (Wheaton *et al.*, 2015). Blanchard (2015) demonstrated that site survey information (consistent with CHaMP habitat sampling but without topography) and a few landscape variables derived from GIS and the 10m DEM could statistically classify with 88% accuracy, reaches delineated through the River Styles framework. Further, she was able to predict the abundance of juvenile steelhead with greater precision and accuracy using reach types delineated using River Styles, network estimates of GPP (which is highly correlated

with temperature), and date, than from the same variables used to classify River Styles (see Figure 4 of Blanchard (2015)). Because we can estimate the same general inputs to NREI models from bottom-up reach level data and top-down “remote-sensed” data (Figure 11), we believe that a mechanistic based empirical model will be able to predict NREI estimates of carrying capacity.

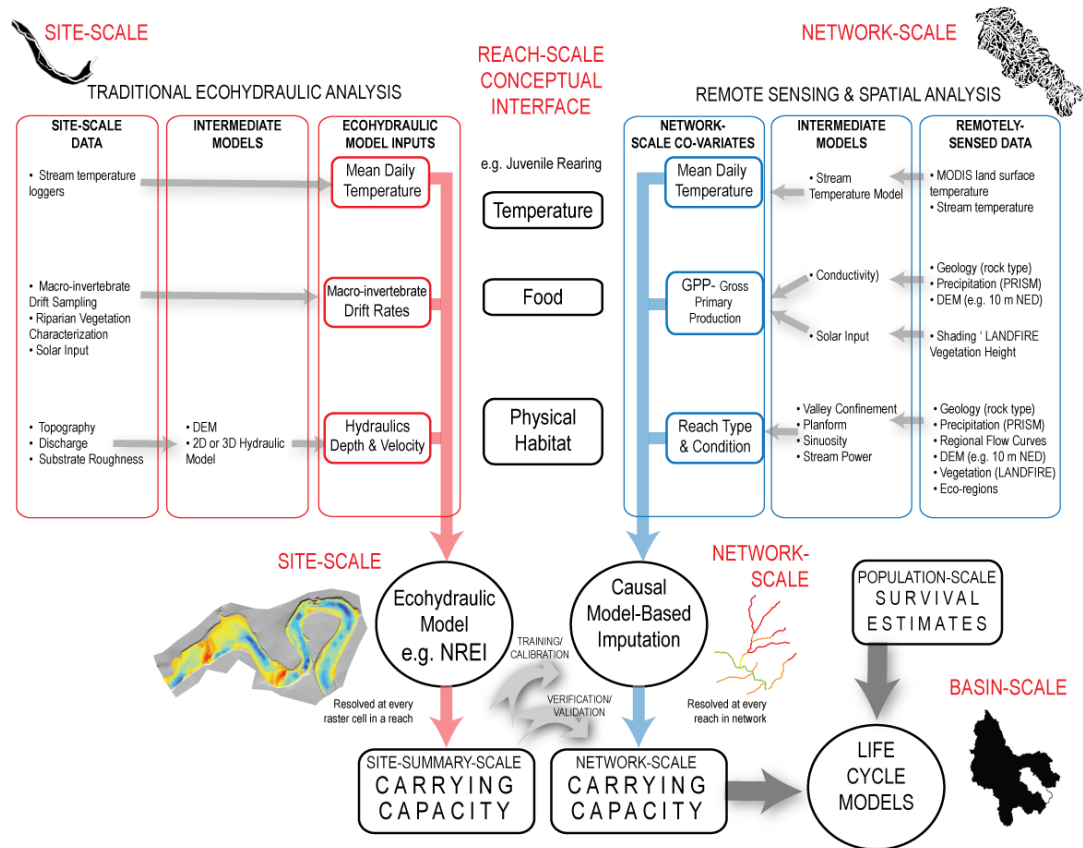


Figure 11- Example of how the components of the proposed framework can fit together for juvenile rearing using NREI model and causal-based imputation to produce robust carrying capacity estimates at the network and population scales to feed life-cycle models. Individual component pieces and concepts can be interchanged. The key attributes are the a) conceptual alignment at the reach-scale between inputs used to drive the site-scale ecohydraulic models and the network co-variates; b) the leveraging of readily available remotely sensed data to support network scale modeling; c) use of traditional site-scale ecohydraulic analysis to train, calibrate causal model-based imputation and ultimately validate it. The framework aims to highlight the analytical tools and underlying theory necessary to transcend spatial scales in the riverscape of relevance to understanding fish population dynamics.

## Population - Life Cycle Modelling

Upon generating watershed-level estimates of capacity for key life stages (e.g. Figure 8D), directly answers to key management questions within a life cycle modelling framework can be explored. A life cycle model (LCM) is simply a mathematical representation of an organism's life history that, given relevant inputs (i.e., stage-specific productivity, capacity, and initial abundance), can be applied in a forward simulation context (Figure 9). The LCM used by CHaMP is a stage-structured (for Pacific salmon and steelhead), stochastic projection model adapted from the 'Shiraz' model developed for similar applications in Puget Sound (Scheuerell *et al.*, 2006). LCMs are particularly well suited for addressing key management questions pertaining to tributary habitat restoration because they simulate future population trajectories as a function of the same demographic parameters that restoration aims to improve, all while explicitly portraying uncertainty in model inputs (Sweka and Wainwright, 2014). LCMs thus offer a means to rigorously tackle planning-related questions like 'will restoration action *X* (or the suite of actions *X*, *Y*, and *Z*) help the population reach a target abundance threshold, within a certain time frame and/or risk tolerance?'; and post-project effectiveness questions like 'could the improvement in tributary survival that's been observed, given survival improvements expected elsewhere (e.g. at mainstem dams), result in population recovery?'. The latter example highlights an important strength of LCMs for assessments involving highly migratory species with diverse life histories and impacts (and restoration measures) occurring at many scales—i.e., stage-specific effects can be assessed amidst an otherwise overwhelming sea of complexity.

In practice, LCMs are parameterized to answer questions pertaining to tributary habitat restoration using up-scaled CHaMP capacity estimates (i.e., juveniles, adult spawners), population-specific estimates of productivity or survival, and levels of variability around these inputs that are commensurate with variation observed in population monitoring data (McHugh *et al.*, In Review). Based on hundreds of Monte Carlo runs spanning a few decades each, simulations from this baseline parameterization give managers a probabilistic view of population performance/status in the absence of intervention across a tractable time horizon. The real power of LCMs in the present setting ultimately traces back to the reach/site scale wherein topographic surveys and ecohydraulics models provide a mechanistic basis for prospective restoration gaming. For instance, CHaMP DEMs can be modified to simulate the

effects of large woody debris placement (e.g. Wall *et al.*, 2016), a go-to restoration treatment for structurally simple salmonid habitats. The benefits of these treatments can then be quantified through hydraulic and NREI model reruns and up-scaled accordingly. Given its bioenergetics origins, NREI can also be used to quantify fisheries benefits arising from anticipated temperature improvements (e.g. resulting from riparian revegetation, ((ODEQ), 2010). Thus, the ability of LCMs to provide meaningful insight on the population-level benefits of restoration relies heavily on a thoughtful integration of information from multiple scales.

## Discussion

### Plurality of Approaches for Component Pieces

The focus of this paper was not on the *how* each component piece of this framework was done, but instead on the broader issue of *what* is needed from each component to facilitate the integration of these component pieces to support life cycle modelling in a meaningful way. Each piece represents the efforts and collective advances of various sub-disciplines of geomorphology, ecology, fisheries biology, hydraulics, and ecohydraulics. In this paper, we have not digressed into the methodological details for each component piece and instead emphasized an overall picture. However, it is important to recognize that many different approaches can be usefully substituted in each step, each with its own costs, advantages and disadvantages and corresponding uncertainties.

For example, the implementation of the framework we have presented here would not be possible without numerous advances in remote sensing (Carbonneau and Piegay, 2012; Gilvear *et al.*, 2016) from the past two decades. There are many different remote sensing techniques that can be combined in various ways to paint a quantitative picture of physical habitat with both topography and aerial imagery (Bangen *et al.*, 2014a). So long as the uncertainties in each approach are adequately represented (Wheaton *et al.*, 2010b; Bangen *et al.*, 2016), and their significance to the analyses and questions at hand appropriately considered (Wheaton *et al.*, 2008), many different approaches should apply. Our premise here was that as long as each component piece had a pragmatic way to produce the component output needed that was scientifically defensible, it would suffice towards

achieving the broader integration and ultimately addressing some pressing key management questions.

Moving forward with application of this framework, we have been exploring tradeoffs and robustness of different methodological approaches to producing outputs for each component. In some cases, simultaneously implementing a plurality of approaches to get alternative 'solutions' may be the best way to consider uncertainties (Rotmans and Van Asselt, 2001). For example, we may have a variety of habitat suitability curves to drive HSI models (Breecher *et al.*, 2016) and varying levels of complexity of biotic or physical process representation in FIS models of spawning habitat suitability (Ahmadi-Nedushan *et al.*, 2006; Hayes *et al.*, 2016). The contrast between spatial predictions of habitat quality vary between models of different degrees of complexity (e.g. Figure 5 for HSI vs Figure 6 for NREI), but it is not yet clear whether this refined mechanistic representation produces markedly different or better estimates of capacity at the site-summary-scale. If different approaches to 'answer' the question of how many fish a reach can support converge on a similar result, some confidence in the robustness of that result is instilled. Conversely, 'different' answers can be combined into a distribution of probable values and reflected with uncertainty around a LCM input (cf. Rotmans and VanAsselt, 1996; Rotmans and Van Asselt, 2001; Sear *et al.*, 2008). Additionally, the extent to which those competing model representations paint a divergent picture also highlights specific localities, issues, processes and factors that deserve closer attention and/or further research. All of the raw data used to drive the models here is already publicly available (<http://champmonitoring.org>), and we are working to make all the model product outputs publicly available in the near future. We encourage other investigators to make their own refinements, model substitutions and alternative analyses.

Why hasn't this already been done? What made it possible?

The framework presented here is not the first attempt to make meaningful use of habitat measurements at a network scale to more accurately estimate capacity (McMillan *et al.*, 2013). However, to our knowledge, it is the most ambitious scope over which such an upscaling attempt has been made and perhaps the only one that attempts to preserve the kind of spatially explicit, reach-level resolution needed to support restoration

planning/implementation. We have focused on pragmatic approaches that are possible to implement today given available remote sensing information, as well as an overall framework that is theoretically sound, and conceptually aligned with the best available science. Benda *et al.* (2016) highlighted that the remotely sensed data (particularly DEMs to derive robust drainage networks) are not equal in all parts of the globe. The data we used to drive our network models is nationally available in the US, and similar or better products exist in some European countries, but worldwide equivalents are not yet available. Though recent breakthroughs in producing 2m DEMs from satellite imagery may improve this in coming years (e.g. Noh and Howat, 2015). Therefore, we hope that the generic aspects of this framework will be transferred to catchments in other parts of the world, other populations and other species as this sort of data becomes more broadly available. We know the framework can be improved, and we do not propose it as a rigid, static manual proposing how to tackle the problem of making reach-scale physical habitat data more useful to fish population modeling.

The reasons, we believe, that such a framework has not already been implemented has mostly to do with the fact that the data requirements and computing power required to implement and scale all of the component pieces described here have simply not been previously available. While various habitat monitoring methods have been in existence for at least thirty years (Kaufmann *et al.*, 1999; EA, 2003; Heitke *et al.*, 2010; Somerville, 2010), none of these stick-and-tape approaches produce continuous, topographic data required to support the sort of ecohydraulic analyses that are the backbone of the proposed framework. As researchers, we just happen to be fortunate enough to be tackling these problems at a time when there is a convergence of management need and interest. That combined with a growing availability of remotely sensed data, increased rapidity and ease with which high resolution topography can be acquired (Passalacqua *et al.*, 2015), and the computing power and scalability via cloud-computing exists to now implement modeling approaches and analytical frameworks that have been in existence for decades across population scales. In short, the ecohydraulic, fisheries and geomorphic communities are maturing to the point that we can start to think about not just illustrating new methodological concepts with one off examples (much as we have in this paper), but we can actually begin to transform and combine operational models into production-level analyses. While such an advance is the cornerstone to management

being able to leverage and implement such science in a decision-support context, it also offers an unparalleled opportunity to test some basic hypotheses about how both these physical systems work as well as how these populations respond to physical habitat.

Network-scale models that inform assessments of salmonid populations already exist. Promising riverscape network analyses to specifically look at implications to salmonid populations by Whited *et al.* (2012), Benda *et al.* (2007), Alvarez-Cabria *et al.* (2016).

A perceived barrier to implementing the framework discussed here could be cost. Advances on the production of network co-variates that could be useful in the upscaling steps has been particularly fruitful over the past decade (e.g. Beechie *et al.*, 2003; Benda *et al.*, 2007; Alber and Piegay, 2011; Belletti *et al.*, 2013; Bizzi *et al.*, 2013; Roux *et al.*, 2014; Schmitt *et al.*, 2014; Macfarlane *et al.*, 2015; Demarchi *et al.*, 2016a). Many of the datasets required to drive such network scale analyses are free and their availability is increasing. It would be tempting to conclude from what we presented here that this approach is only possible with massive financial investments in habitat sampling, monitoring, and research. However, there are at least three counter points to such an argument.

First, while CHaMP is attempting to implement these methods across dozens of watersheds in the CRB simultaneously, the sampling effort required to study a single population in a watershed of interest is relatively modest (e.g. circa 25 sites of sampling effort per year) and is likely in the \$50K to \$100K US range for annual sampling effort. We contend that the easy advances remote sensing was going to provide for better characterizing instream habitat have already been realized (Bangen *et al.*, 2014a). There is a basic level of sampling effort required to acquire the necessary topographic data and only minor efficiencies are likely to be realized in the future.

Secondly, with all of the component pieces described in this framework, we have invested heavily in transferability. Specifically, we focus on operationalizing established methods wherever possible, only creating new ones where necessary, and developing protocols (e.g. <http://monitringmethods.org>) and writing software to implement both that are all open-source, transparent, free to use and highly scalable. That is, the analytical cost barriers to

implementing such a framework are quickly diminishing, and the cost of cloud computing is now remarkably low. As an anecdote, we have spent less than \$1000 on computation time running *all* of the tens of thousands of hydraulic model simulations described here. Thanks to Delft making their source code open source, a problem that a few years ago was insurmountable is now readily affordable. Traditional model parallelization requires substantial manual configuration and refactoring of code that is not needed when using virtual computing clouds. The emergence of this cheaper and more accessible parallelization makes these virtual computing services attractive. Virtual computing services currently price their infrastructure rental costs to commercial applications (such as media streaming and desktop cloud software services), which dwarf the size of the sort of research computing needs described in this framework.

Finally, it may be possible to implement some of these approaches across the network based *only* on freely-available remotely sensed data and without large on-the-ground monitoring investments (Macfarlane *et al.*, 2015). However, this is likely only to be the case if robust fish-habitat relationships can be established in other, *similar*, riverscapes. The methodological question we cannot yet address is how robust the fish habitat relationships established empirically and theoretically, here, hold up across locations without such intensive sampling efforts. This may be just wishful thinking, but if it proves possible with reasonable accuracy, it could save fisheries managers millions.

### Implications for Population-level Assessments & Life Cycle Modelling

One of the problems with the ecohydraulic approach is its premise and guiding principles are based on the assumption that the physical environment in riverscapes (primarily as manifested via hydraulic processes, but also geomorphic processes), is a direct driver of ecology. While there is merit and truth, for example, in assuming fish selectively occupy habitat, other biotic factors such as competition, predation, disease, social factors, density dependence, and food availability all can act to modulate fish–habitat relationships (Rosenfeld, 2003). Accordingly, purely physical habitat-based approaches may not always provide useful insight on population limitation in every system (Railsback, 2016). However, in cases where habitat is limiting,



this framework provides a workflow for identifying exactly where and to what extent habitat-related constraints (i.e., diminished capacity) may be limiting a population. There are other promising alternative perspectives on estimating populations emerging, which are not based on full, simulation, life cycle modeling. These include geostatistical upscaling techniques like Isaak *et al.* (2016) that use block kriging interpolation across spatial-stream-networks (SSN; i.e. what we term drainage network here) and fish density surveys. In principle, such approaches could substitute the ecohydraulic analyses described here for fish density surveys in their approach and use their upscaling techniques to assess populations. Similarly, it is possible to use actual fish density surveys in place of the ecohydraulic modeling efforts described here to drive life cycle models to make estimates of current populations. The problem with all three of these alternatives is that they are useful for assessing what a population looks like currently, but do not help for producing realistic and mechanistic scenarios of how restoration might influence future populations. For these reasons, we are pursuing more mechanistic linkages in the framework described here. However, all of these different approaches offer complimentary perspectives on similar problems. Managers will benefit from lots of creative approaches to think about the same problems.

There are several other aspects of the framework presented here that we think make it compelling. First, one of the ecohydraulic models we use, NREI, explicitly considers both a key biotic influence (i.e., food availability) and a primary physiological driver (i.e., temperature) on reach-scale capacity. The FIS-based habitat modelling framework also offers flexibility for incorporating additional variables, such as predation refugia (Muñoz-Mas *et al.*, 2016), which offers additional complexity/realism. Secondly, quantifying capacity is a fundamental parameter in most life-cycle models and can at least help contextualize the question and extent to which habitat may or may not make a difference to populations. Third, the life cycle modelling approach is explicitly driven by a mix of empirical (e.g. survival estimates) and mechanistic fish-centric processes. Moreover, life-cycle models are explicitly considering all stages of an anadromous fish's life.

## **Conclusion**

The multi-disciplinary study of ecohydraulics owes much of its existence to applied management questions surrounding the sustainable management of salmonids and recovery of

their populations. Despite this history, most ecohydraulic studies have been focused at relatively local reach-scales. Advances in remote sensing and computing power have facilitated a more precise characterization of physical habitat for fish and better mechanistic explanation of fish habitat relationships at reach-scales. However, we presented a framework for addressing applied management questions at the broader scale they are typically motivated by —population health and population numbers. The population-scale is not a scale at which traditional ecohydraulic inquiry typically operates.

The crux of this framework hinges around reliable upscaling of our reach-scale understanding of ecohydraulic fish-habitat relationships to the network-scale at which fish populations operate. We described the many component pieces required to facilitate this integration and alluded to the insights into population dynamics it can provide, using examples from the Columbia River Basin (CRB) and the CHaMP (Columbia Habitat Monitoring Program). The CHaMP program captures an unprecedented amount of detail on fish habitat at over 900 sites in 12 sub-watersheds of the CRB. We argue that the primary outputs needed to better inform life-cycle modeling efforts and underscore the significance of tributary habitat in the freshwater life stages of anadromous salmonids were more reliable estimates of capacity (i.e. the upper limits on the number of fish the habitat can support). However interesting such estimates may be at individual sampled reaches, for them to be useful to understanding their impact on population dynamics, these estimates need to be made at the scale that the population exists – i.e. across the entire drainage network.

To reliably extrapolate estimates of capacity across entire drainage networks (i.e. watershed extents and reach-scale resolution), we suggest two primary requisites must be met. First, the detailed sampling and ecohydraulic analyses made at sites must be done with a survey design that places sampling effort at sites in a spatially balanced, randomly selected, but geomorphically stratified manner to allow robust statistical inference and upscaling. Secondly, the extrapolation to the network needs to be done with spatial co-variables that help explain physically and biotically why fish habitat and subsequent estimates of capacity vary systematically across a drainage network. For our examples of estimating juvenile rearing capacity and spawning capacity, we suggest that geomorphic reach type and condition variants are required for both. However, for juvenile rearing capacity, we suggest that stream

temperature (as a determinant of metabolic rate) and proxy for food availability (primary production) are additionally useful. These spatial co-variables need to be realistically measurable or modelable at network extents and should correlate strongly to measures that can be made at individual sites. It is only because of advances in remote sensing yielding spatially continuous models of these co-variables that we can even begin to consider how to put these various pieces together in a quantitative and conceptually coherent framework to directly address key management questions about populations. All of the component pieces can be substituted out with more refined, more precise and improved methods and models as and if appropriate. However, before investing too heavily in individual component pieces, we think the next step is to see how sensitive life-cycle model results are to different aspects of the tributary habitat capacity estimates. In some instances, in some watersheds, a simple exploration of the full range of habitat capacity conditions and maximum plausible improvements possible with restoration may reveal that no detectable improvements toward population targets are possible. By contrast, in other watersheds, this framework could be very powerful for helping identify precisely where in the drainage network habitat restoration could lead to capacity improvements that actually can facilitate positive population level responses. For the future, we suggest that the framework could be used to strategically develop more realistic expectations beyond the optimism and wishful thinking that underlie too many restoration actions today.

## Acknowledgements

The lead author (JMW) is grateful to the guest editors David Milan, George Heritage and Neil Entwistle of this special issue and conveners of the 'Remote sensing applications for hydro and morphodynamic monitoring & modelling' session at the 11<sup>th</sup> International Symposium on Ecohydraulics, for both their invitation to present and extreme patience and encouragement with my delinquent submission of the manuscript. This study was made possible thanks to the long term funding investment by Bonneville Power Administration (BPA Awards: CHaMP 2011-006 and ISEMP 2013-017) through the Integrated Status and Effectiveness Monitoring Program (ISEMP: <http://isemp.org>), Columbia Habitat Monitoring Program (CHaMP: <http://champmonitoring.org>), and NOAA Fisheries. Sub-awards from Eco Logical Research to JMW at Utah State University (USU Grant Awards: 100652, 150737, and 200239) supported six of the coauthors involvement in this research. We are grateful to the over 20+ field crews who have worked so hard in the field to collect and post process the habitat data over the past five years. The scope of what is presented here is only possible because of the investment and commitment of CHaMP partners and contractors, including: Oregon Department of Fish and Wildlife, Columbia River Inter-Tribal Fisheries Commission, Shooshone Bannock Tribes, South Fork Research, NOAA Northwest Fisheries Science Center, Quantitative Consultants, Inc. (Claire McGrath and Chris Beasley), Terraqua (Stephen Fortney), Watershed Solutions (Boyd Bouwes and Jeramiah Hittake), North Arrow Research (Matt Reimer), and SITKA Technologies. None of this is possible without the leadership and organization of the CHaMP Management team, in particular Mike Ward (Terraqua), Boyd Bouwes (Watershed Solutions), Sara Walker and Pamela Niele (Terraqua), as well as our Bonneville Power Administration contracting officers – David Burns and Rick Golden. The Ecogeomorphology and Topographic Analysis Lab at Utah State University has been a critical part of the team behind this effort and we are grateful to Wally Macfarlane, Josh Gilbert, Jordan Gilbert, Gary O'Brien, Martha Jensen, Elijah Portugal, Kenny DeMeurichy for their GIS, spatial analysis, surveying and programming support. We are particularly grateful to Chalese Hafen and Martha Jensen for their cartographic support and producing some of the figures in this manuscript. Finally, we thank Sean Welch (Bonneville Power Administration) for his help in encouraging a habitat-monitoring program more specifically on key management questions, which forced us to think about how we could link

the reach-scale monitoring and ecohydraulic analyses to the network-scale over which these populations exist. We also thank two anonymous reviewers for their time and insights, which helped improve the clarity of the manuscript. The views and opinions expressed in this manuscript do not necessarily reflect those of our project partners, their organizations or our employers.

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SPATIAL SCALE OF RIVERSCAPE ANALYSIS

Increasing  
Low

Broad

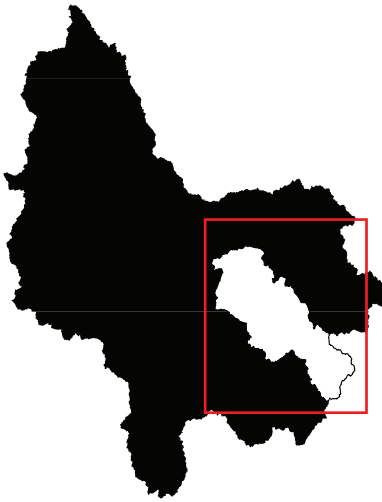
REGIONAL-SCALE



SIGNIFICANCE TO FISH

Multiple Populations

BASIN-SCALE & WATERSHED-SCALE

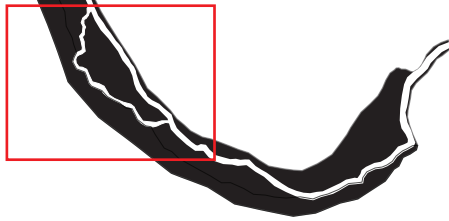


NETWORK-SCALE



Individual Population

SITE-SCALE



GEOMORPHIC UNIT-SCALE

Multiple Individuals

Moderate

Large

High

Small

RESOLUTION

EXTENT

Geomorphic Units

- Run
- Terrace
- Pool
- Channel Cutoffs
- Point Bar
- Bar

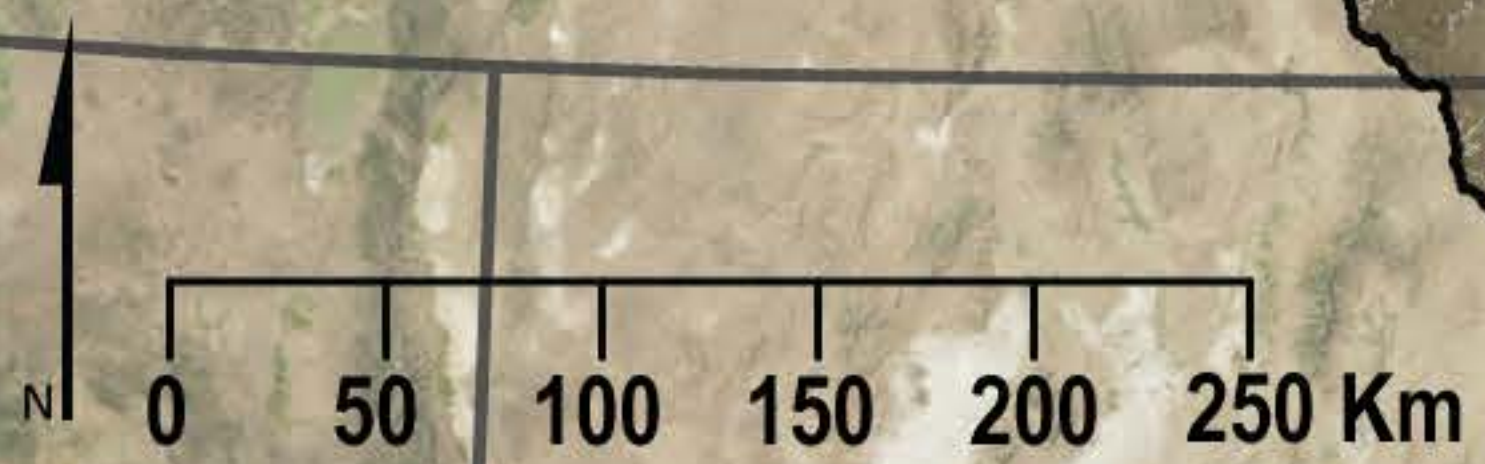




# Columbia River Basin North America



- Perennial stream network
- Chinook extent
- Steelhead extent
- Combined chinook and steelhead extent
- Fish barriers (man-made)
- Major dams
- CHaMP watersheds



CANADA  
UNITED STATES OF AMERICA

WASHINGTON

MONTANA

OREGON

IDAHO

Columbia River

Snake River

Columbia River

Snake River



MONITORING

e.g. ISEMP

e.g. CHaMP

e.g. AEM



Site: CHaMP Survey

**LEVERAGING**  
Existing, Free, CB-wide Data

**?**

**INTEGRATION**  
of Complementary Program Data

ADAPTIVE MGT: MONITORING EFFICIENCY ADJUSTMENTS (CHaMP/ISEMP/AEM)

ADAPTIVE MGT: IMPLEMENTATION DRIVEN ADJUSTMENTS (CHaMP/ISEMP/AEM)

Author Manuscript

KMQ 1  
STATUS & TREND  
What are the tributary habitat limiting factors impacting the "HEALTH" of listed salmonids?  
RPA 56

PLANNING, PRIORITIZATION, & IMPLEMENTATION  
KMQ 2  
What tributary IMPROVEMENT ACTIONS would provide the most freshwater production benefits to listed salmonids?  
RPA 57

ACTION EFFECTIVENESS & ADAPTIVE MANAGEMENT  
KMQ 3  
How EFFECTIVE are tributary improvement actions at providing benefits to listed salmonids?  
RPA 34 & 35

CONDITION MAPS

LIMITING FACTORS MAPS

RECOVERY POTENTIAL MAPS

STRATEGIC TRIBUTARY HABITAT IMPROVEMENT PLANS

PROJECT HABITAT IMPROVEMENT DESIGNS

TRIBUTARY IMPROVEMENT ACTIONS

ACTION EFFECTIVENESS MAPS

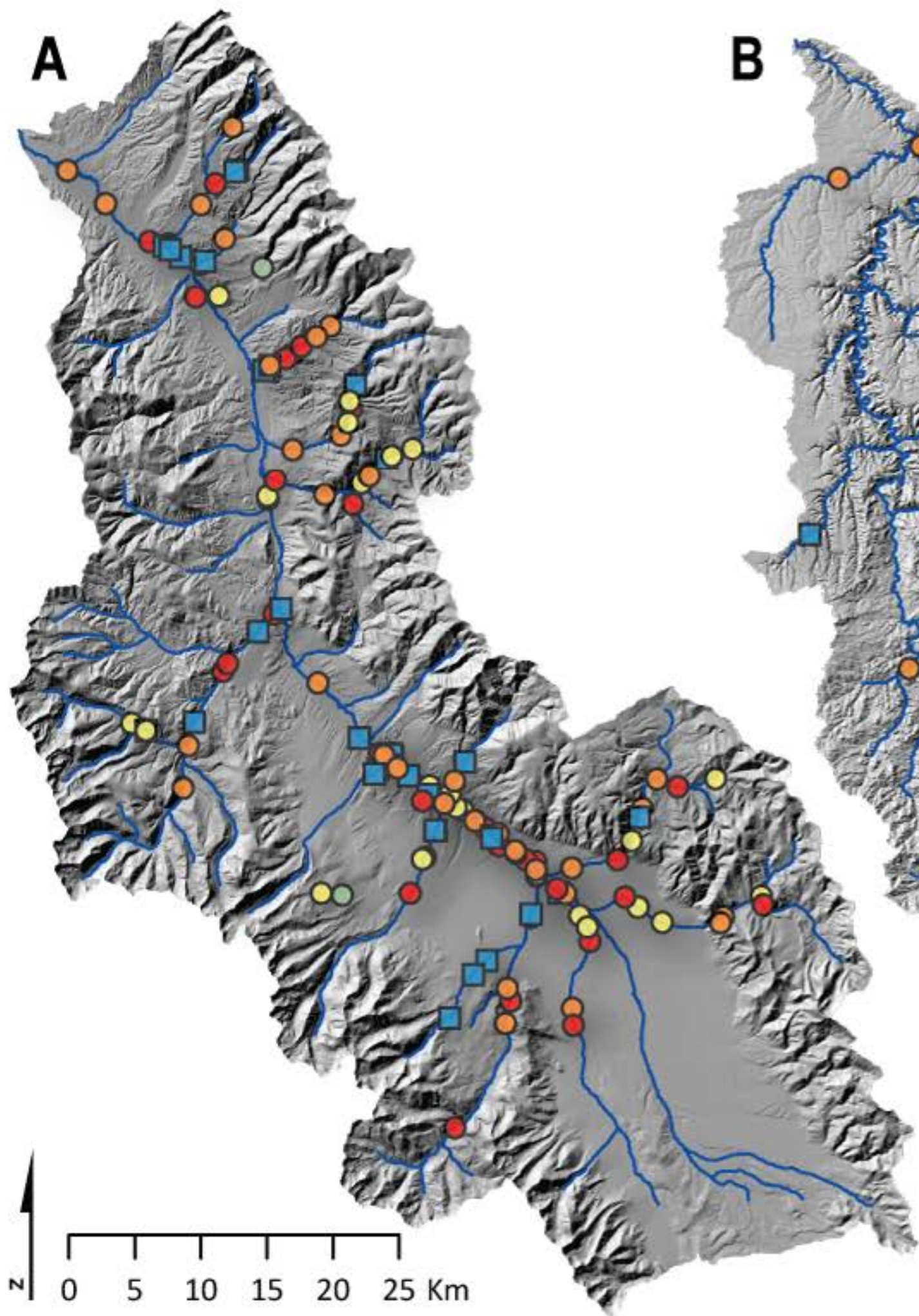
ASSESSMENT

PLANNING

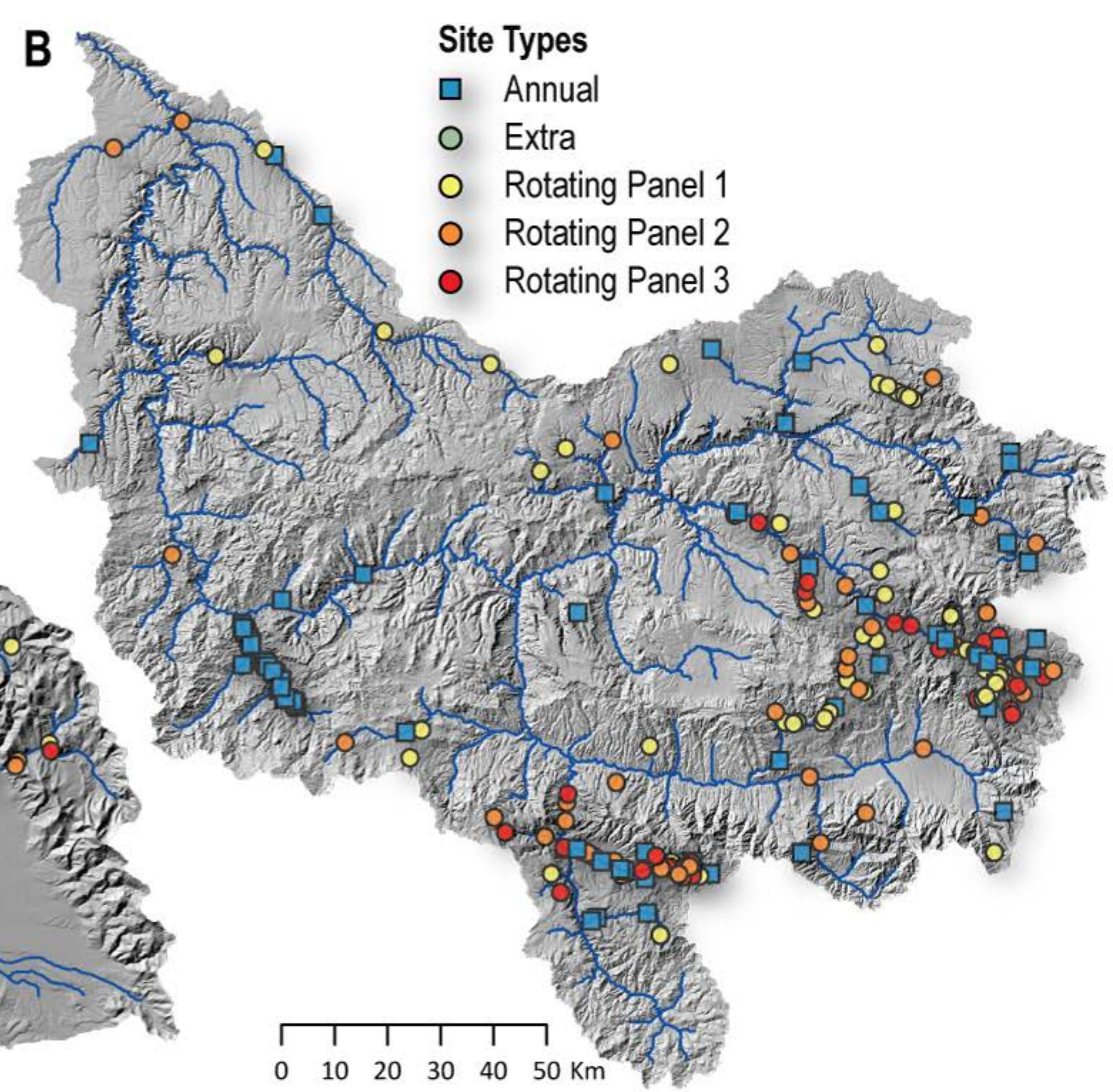
DOING

EVALUATION





80 Annual [200 Total] Sites



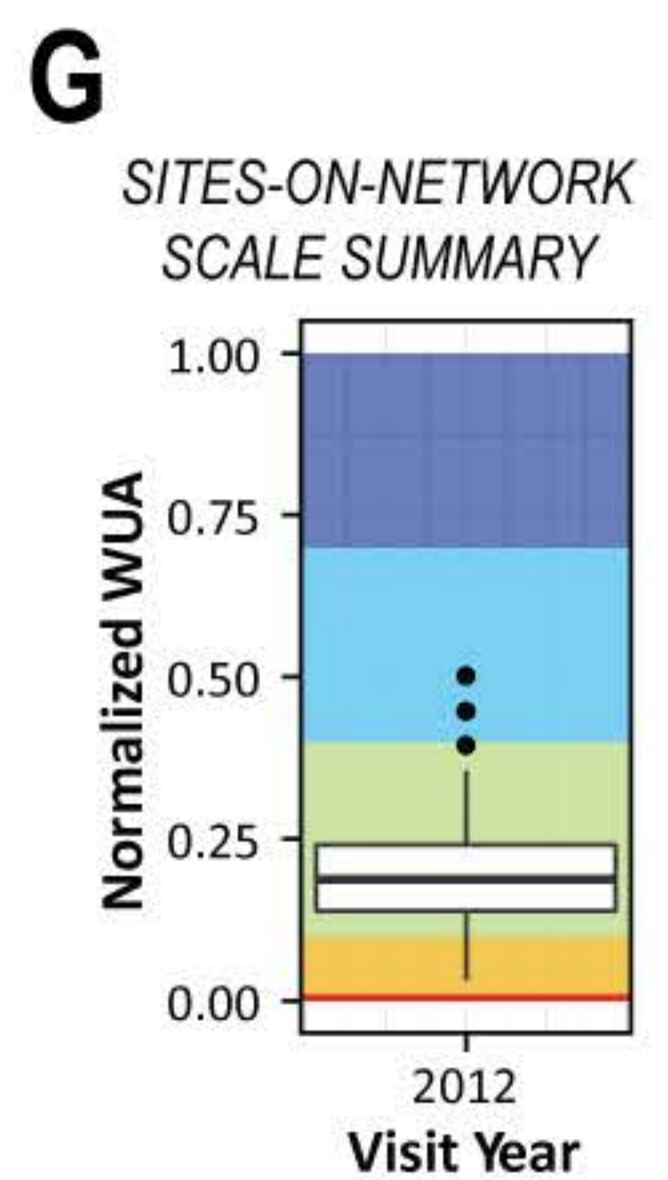
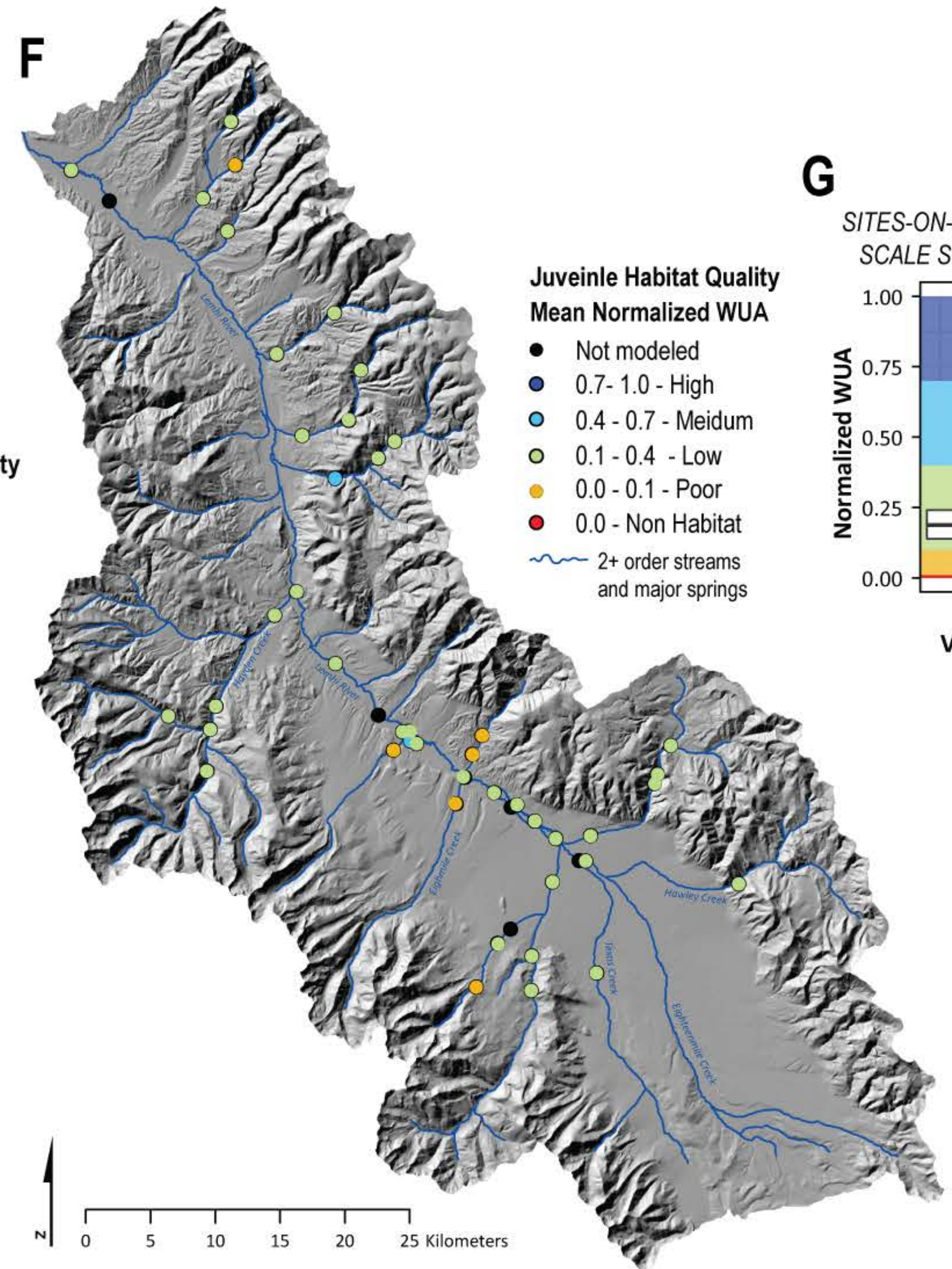
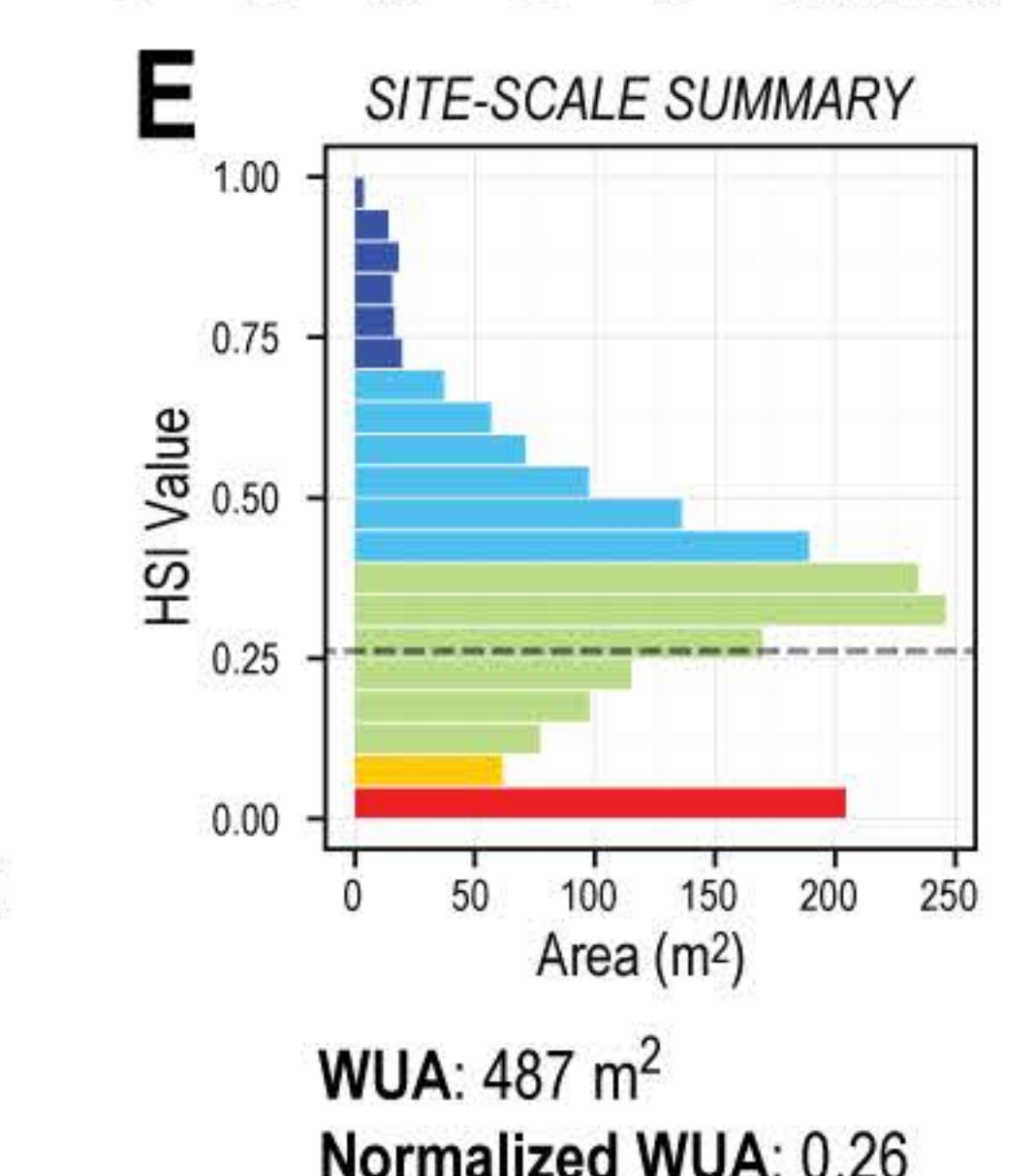
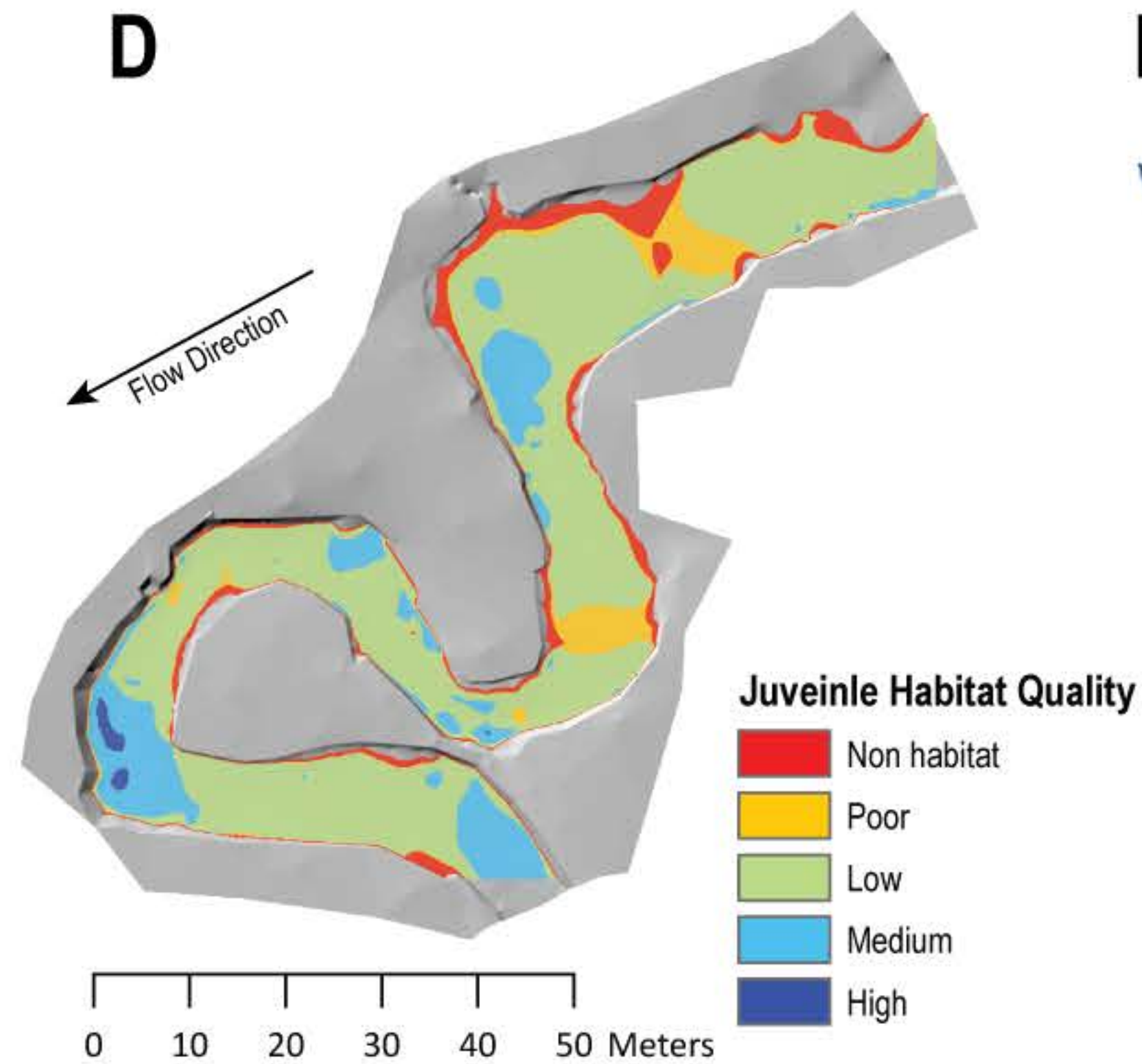
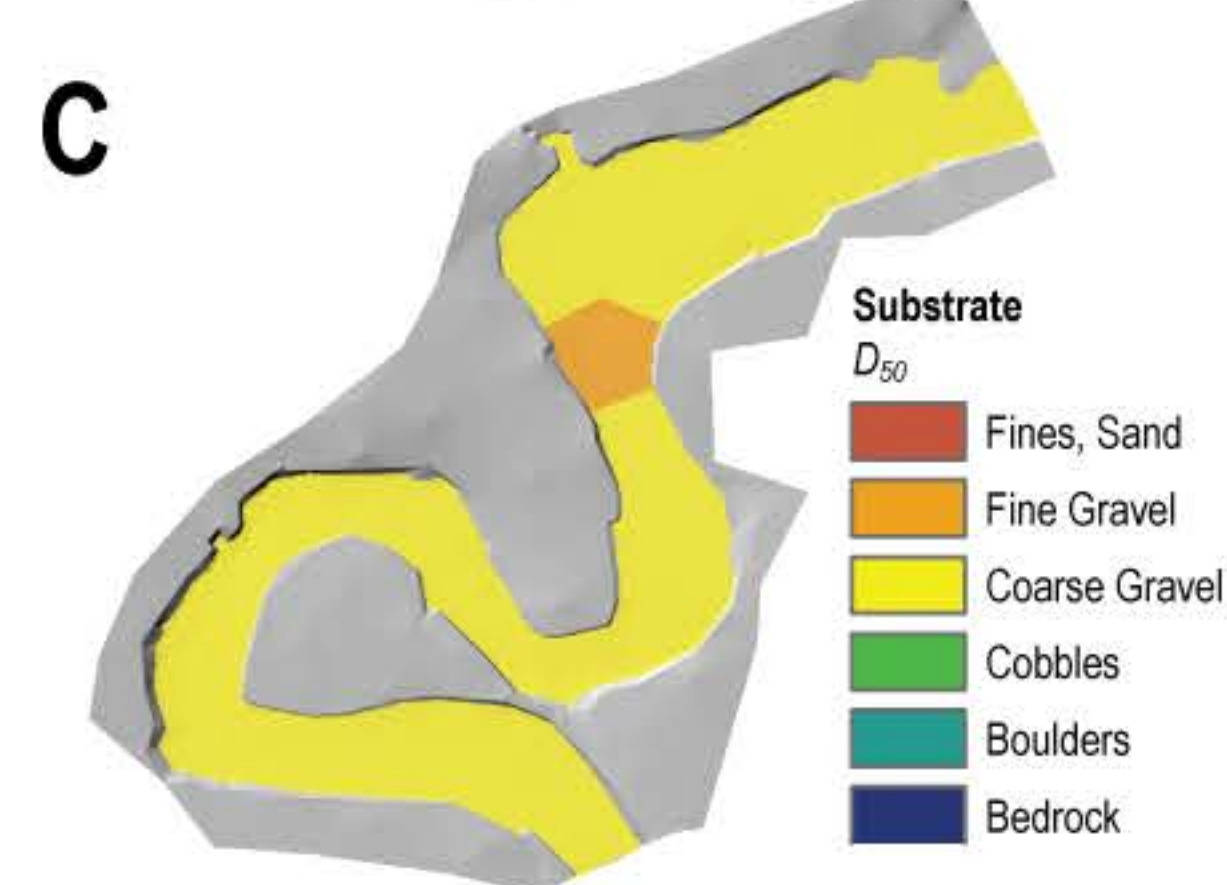
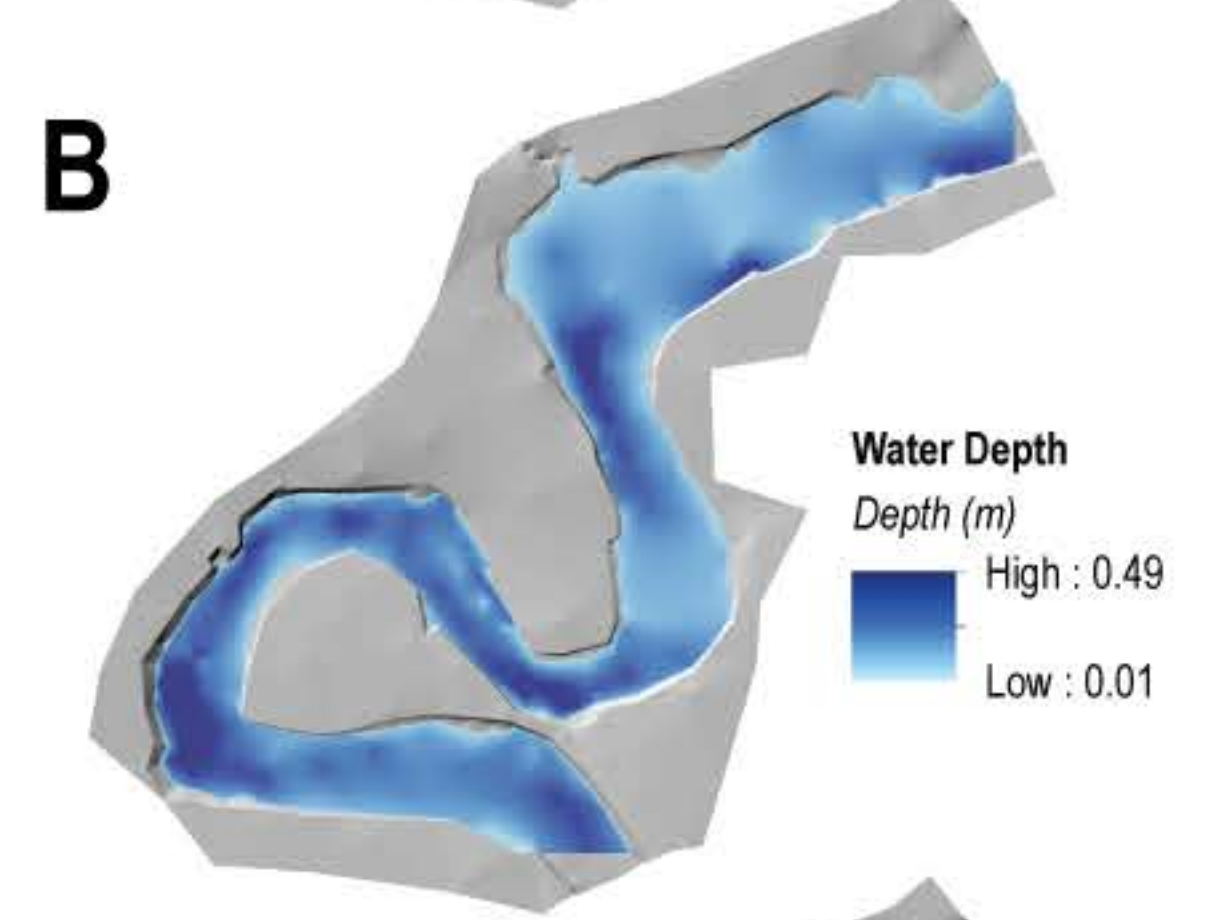
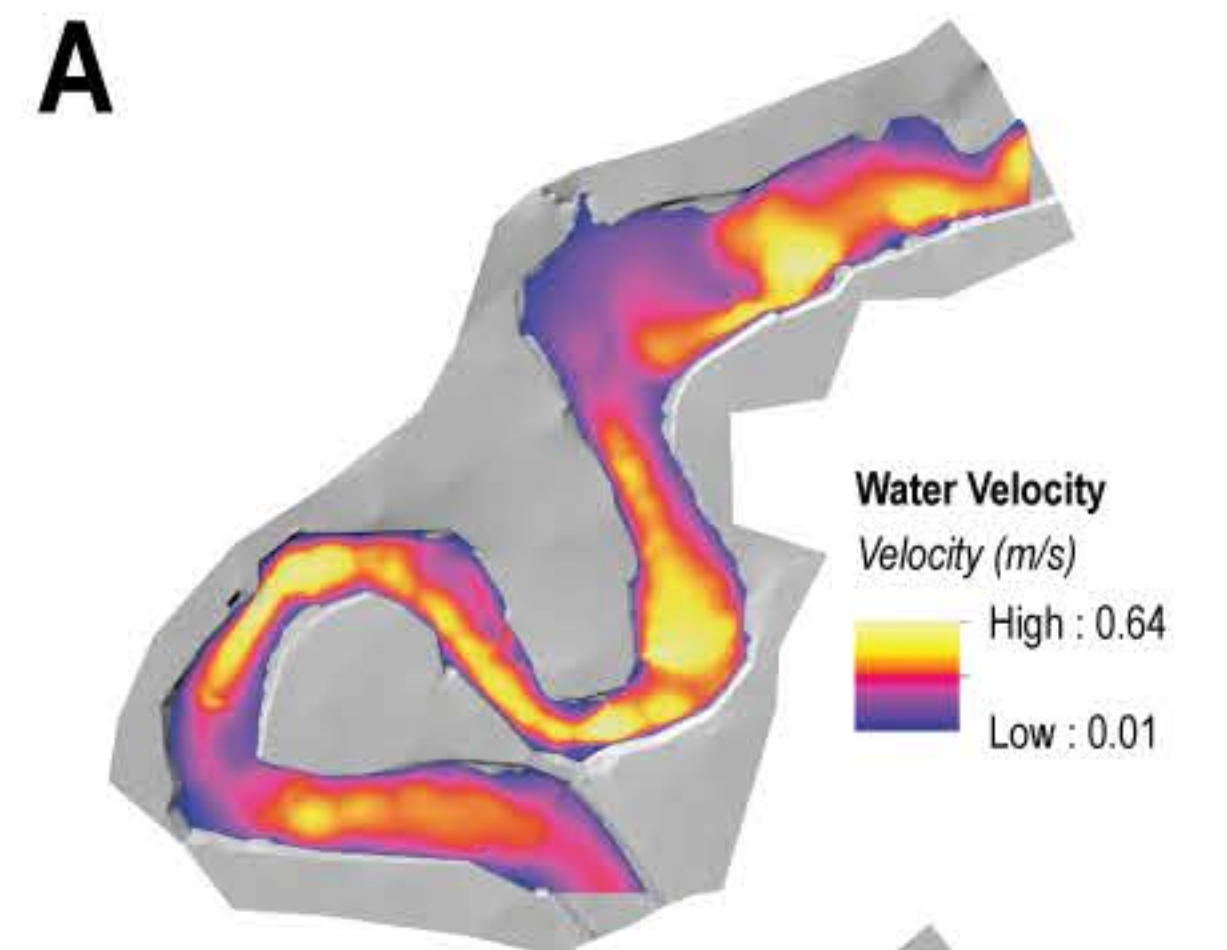
196 Annual [376 Total] Sites



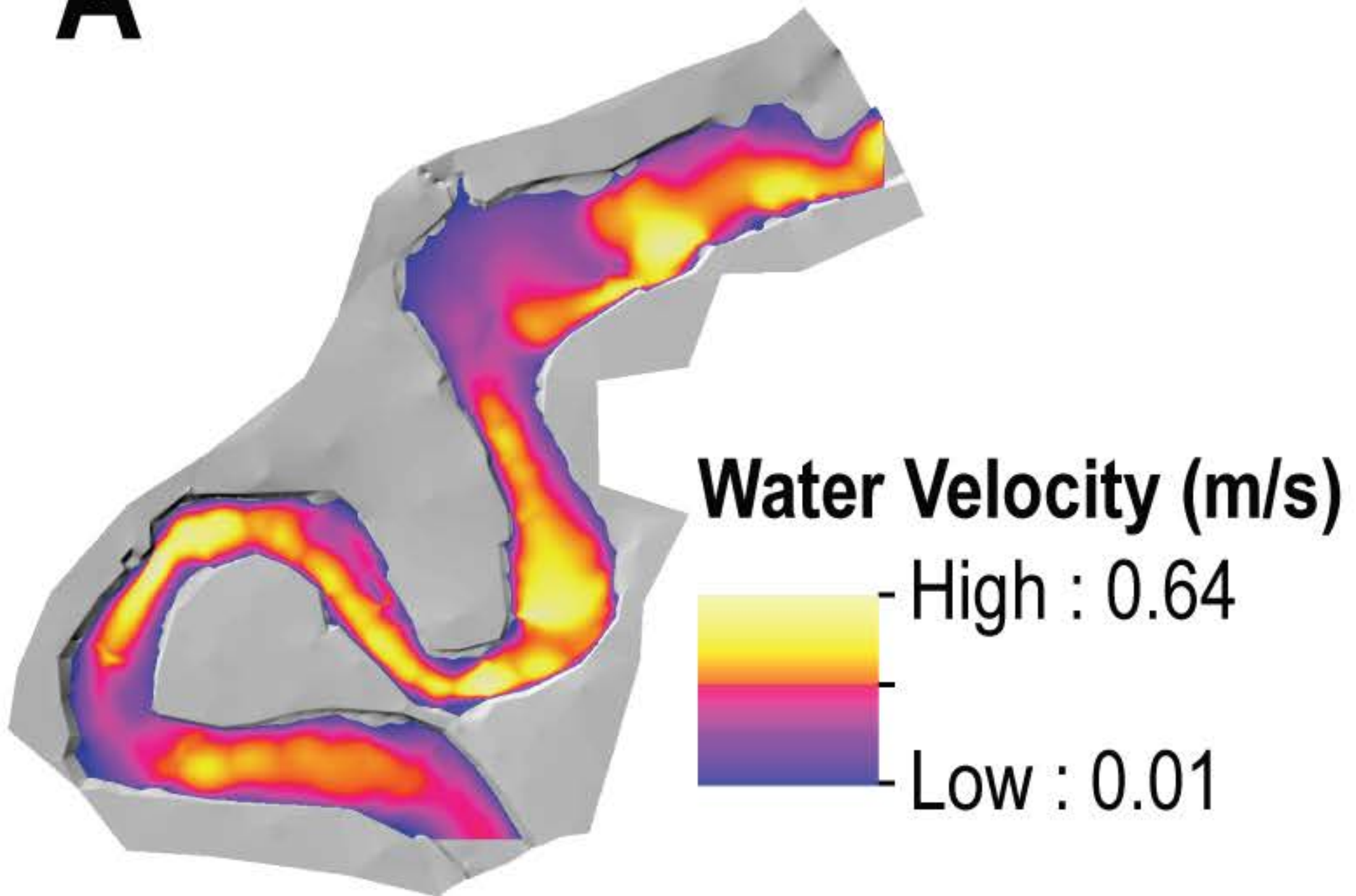
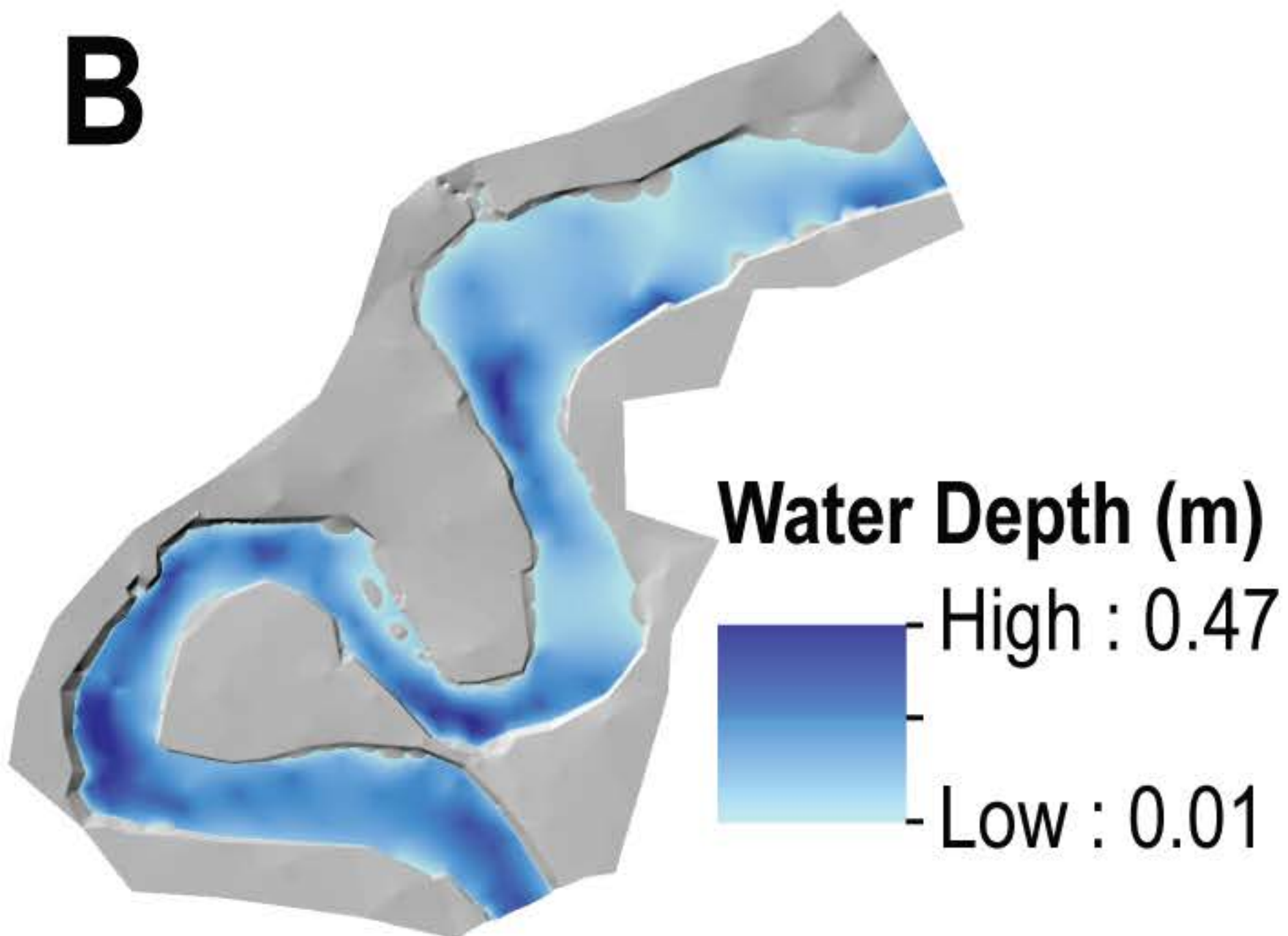
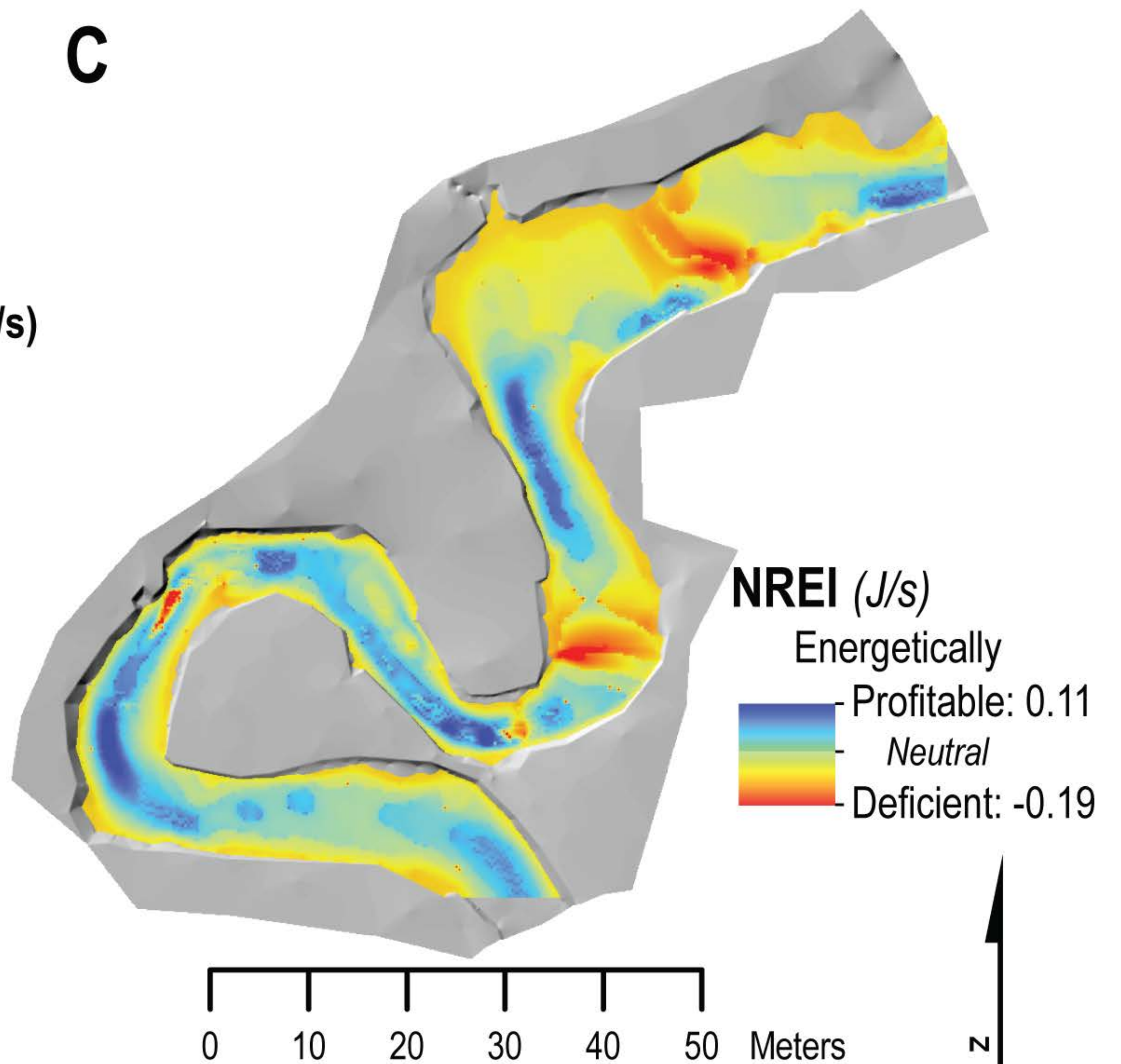
# Chinook Juvenile, 2012 Primary Visits

*REACH-SCALE [Big Springs, Lemhi Watershed, ID]*

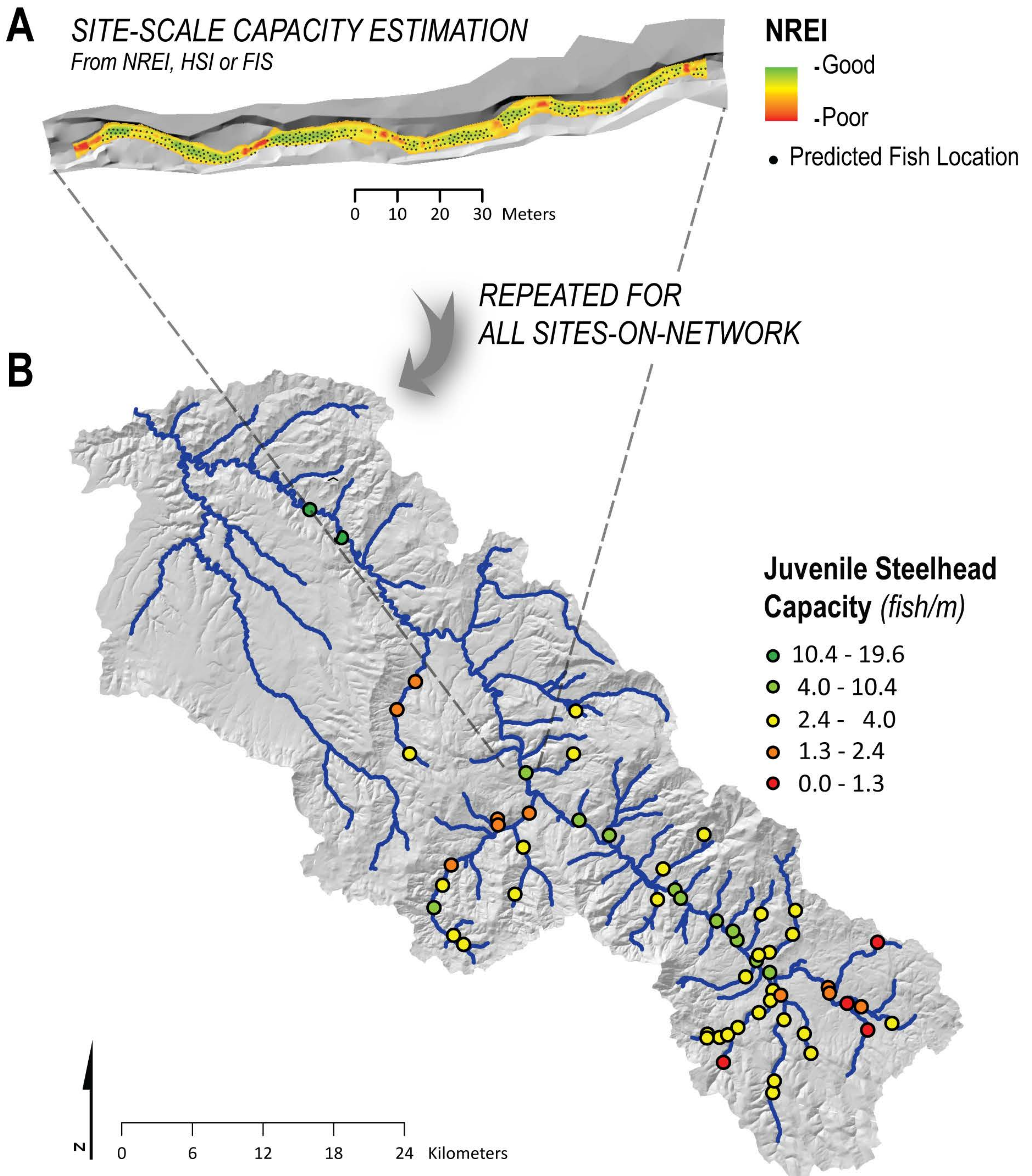
*SITES-ON-NETWORK SCALE [Lemhi Watershed, ID]*



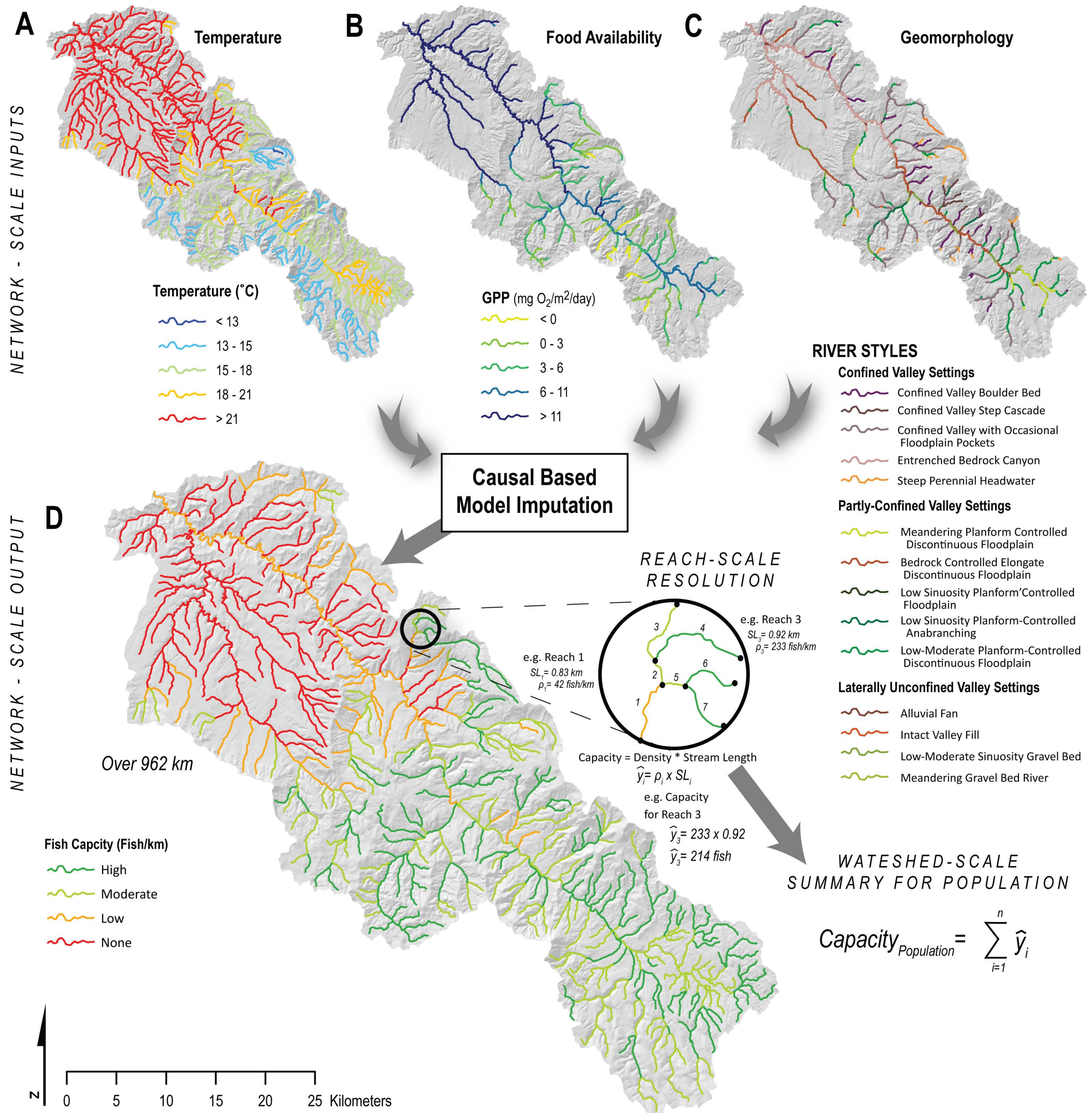


**A****B****C**









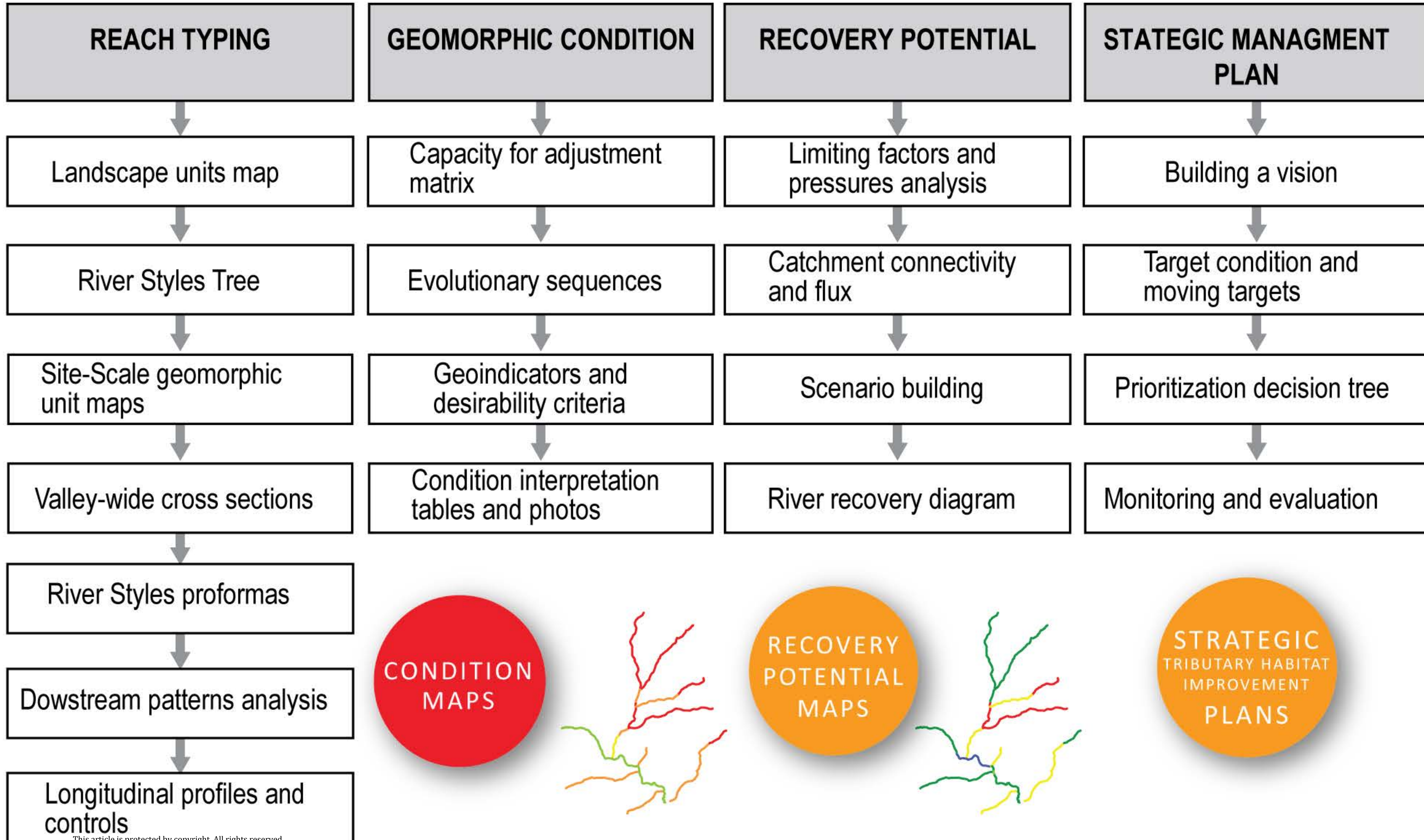


*Used as Network Co-Variates*

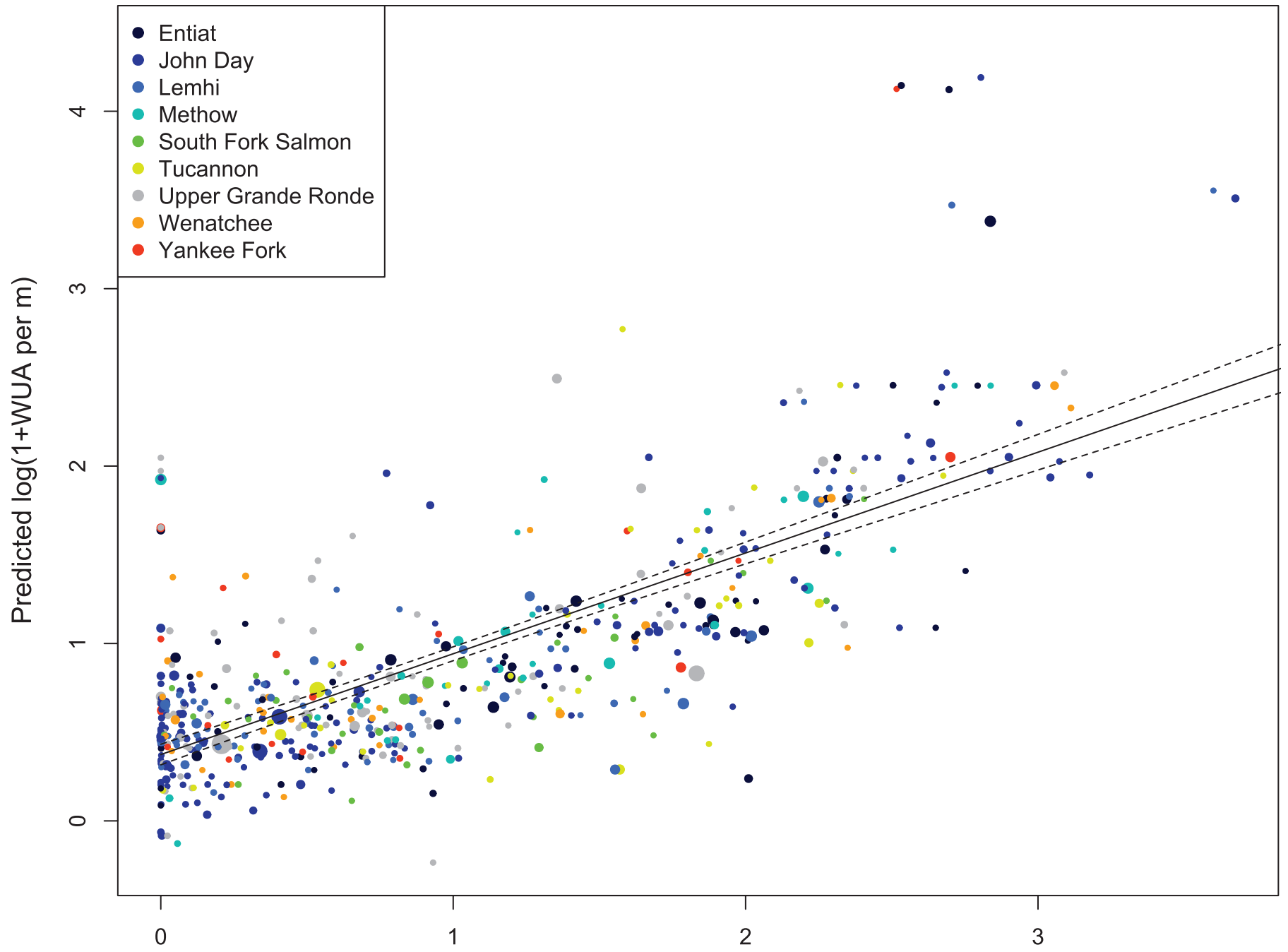
**KMQ1**

*Used to Build Scenarios of Plausible Condition*

**KMQ2**



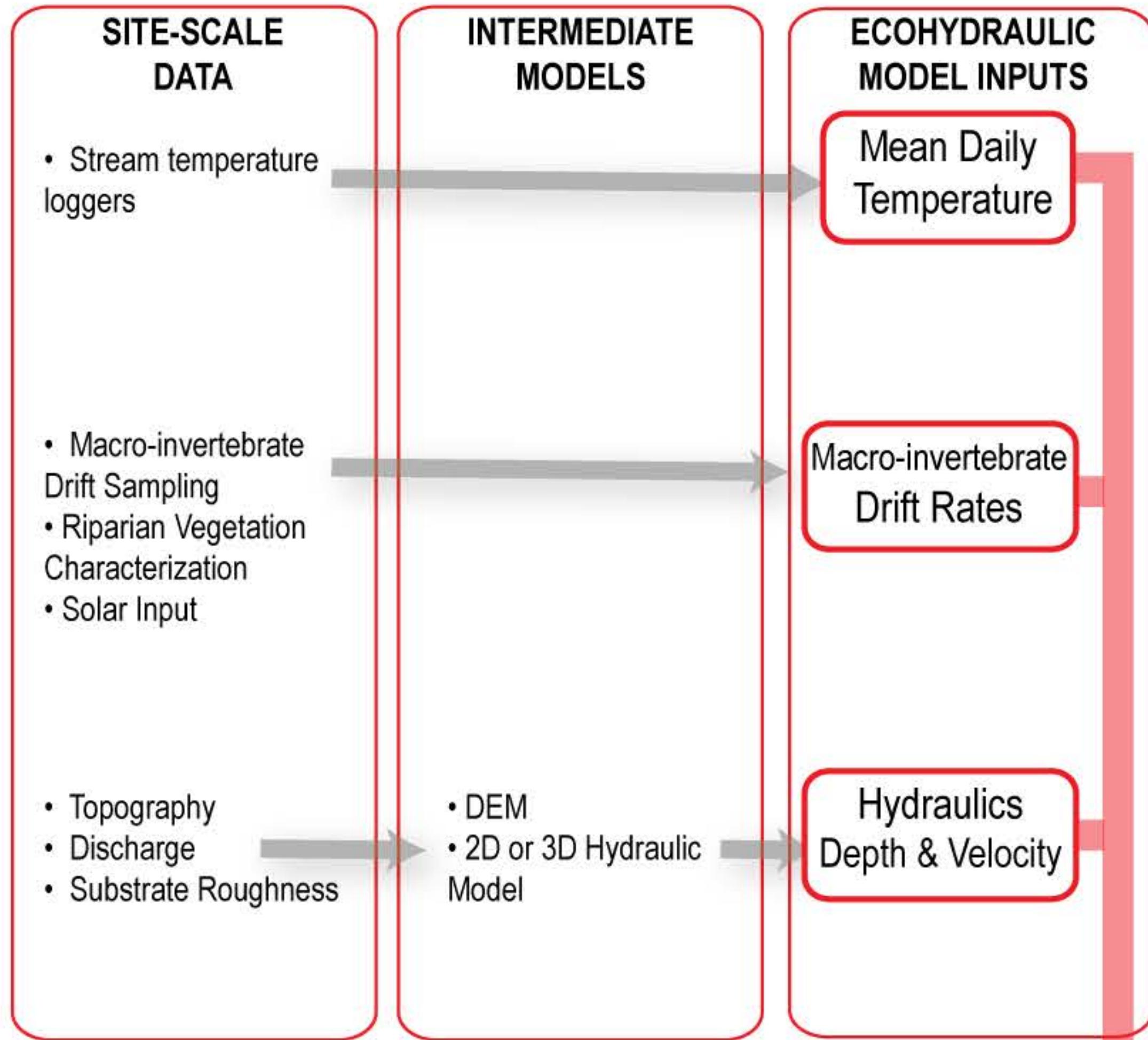






# SITE-SCALE

## TRADITIONAL ECOHYDRAULIC ANALYSIS



# REACH-SCALE CONCEPTUAL INTERFACE

e.g. Juvenile Rearing

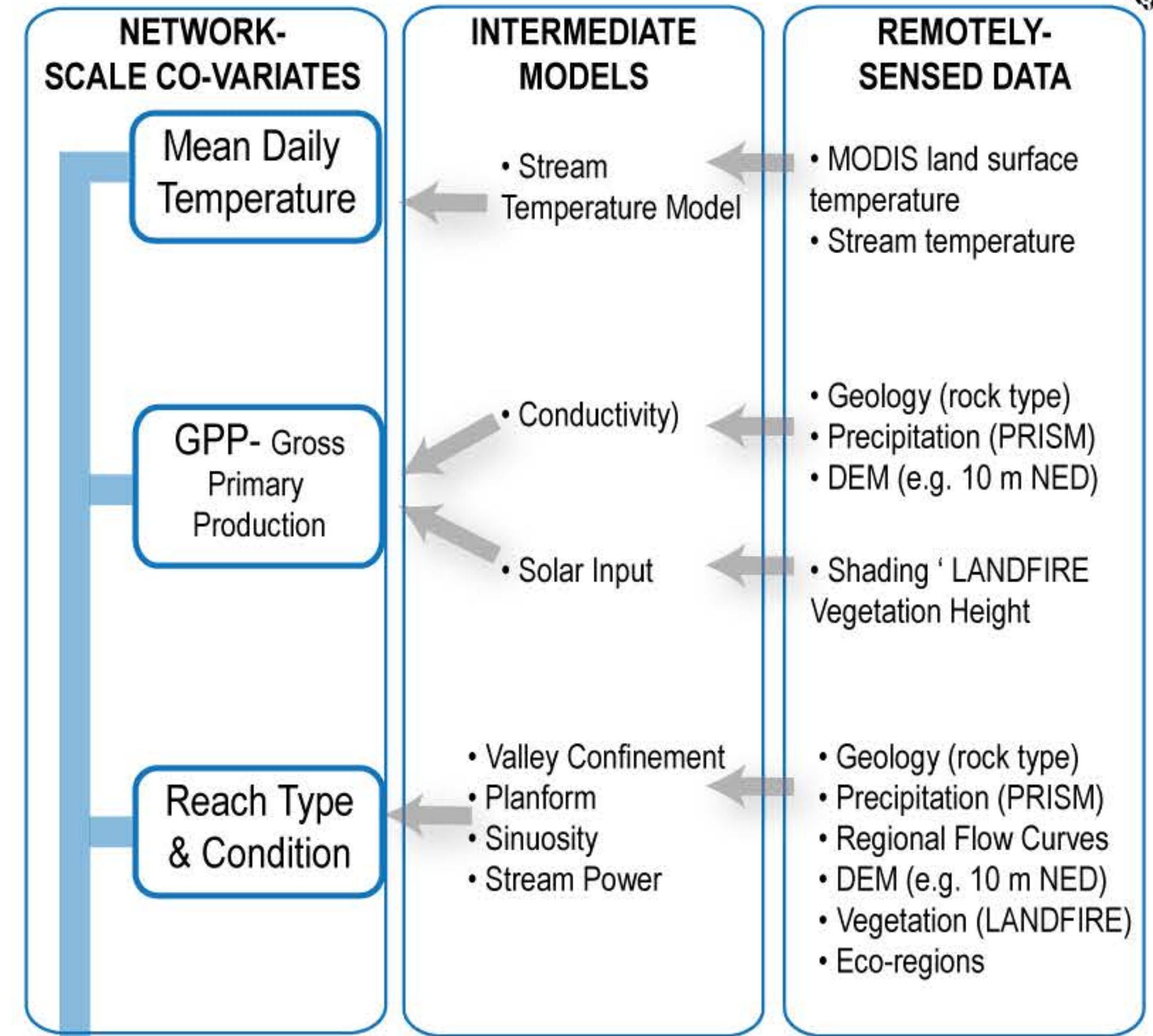
Temperature

Food

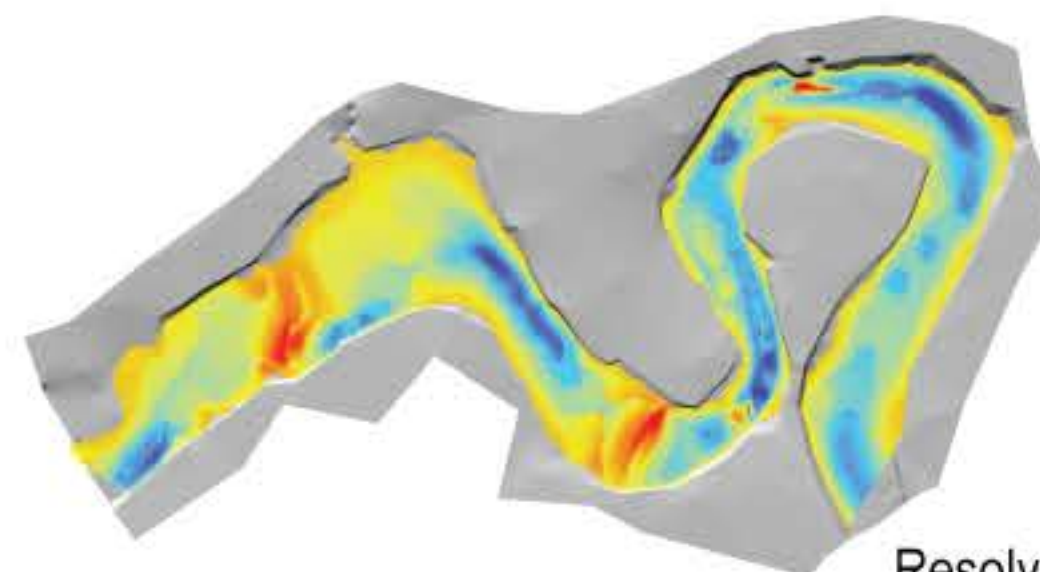
Physical Habitat

# NETWORK-SCALE

## REMOTE SENSING & SPATIAL ANALYSIS



# SITE-SCALE



Resolved at every raster cell in a reach

Ecohydraulic Model  
e.g. NREI

TRAINING/  
CALIBRATION

VERIFICATION/  
VALIDATION

SITE-SUMMARY-SCALE  
CARRYING  
CAPACITY

# NETWORK-SCALE



Resolved at every reach in network

Causal Model-Based Imputation

NETWORK-SCALE  
CARRYING  
CAPACITY

POPULATION-SCALE  
SURVIVAL  
ESTIMATES

LIFE  
CYCLE  
MODELS

# BASIN-SCALE

