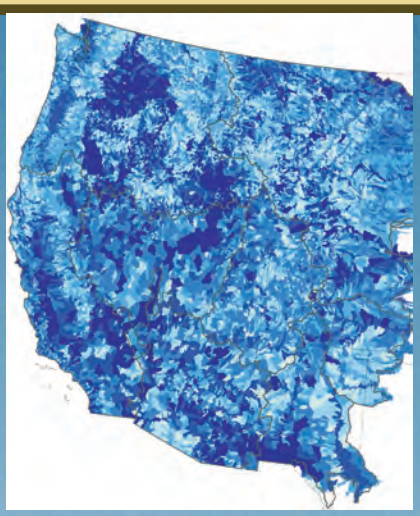

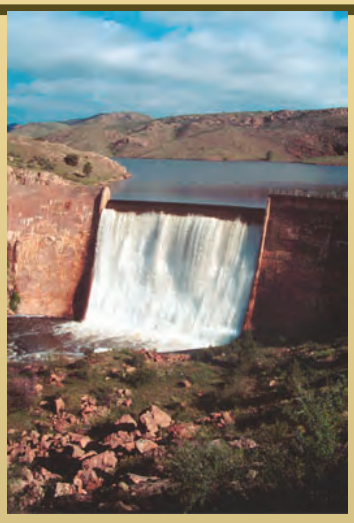

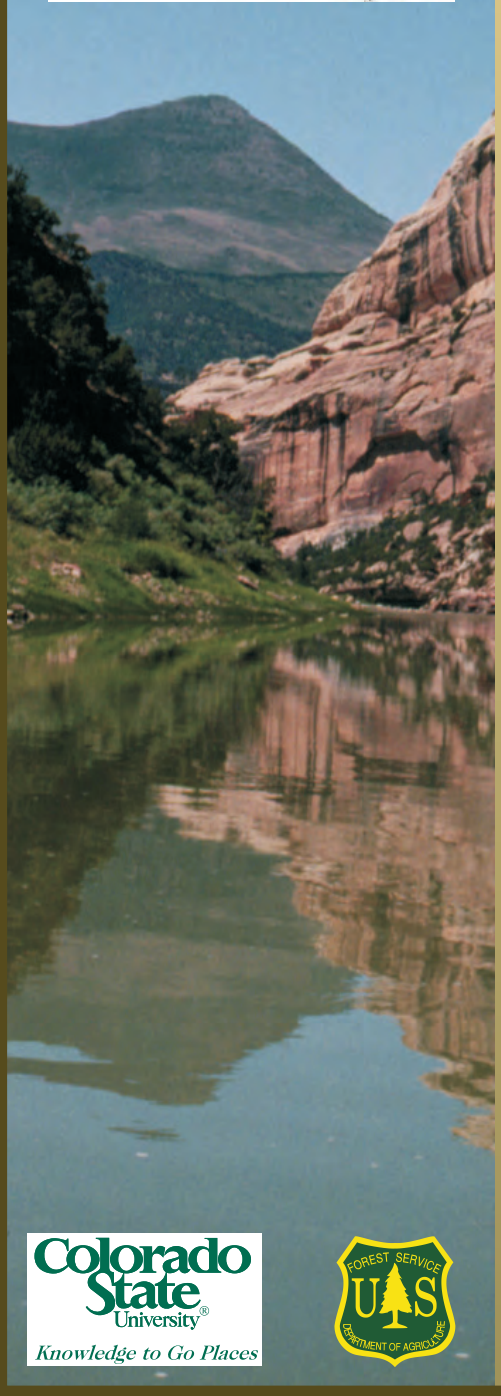

Assessment of Threats to Riparian Ecosystems in the Western U.S.

Prepared for the Western Environmental Threats Assessment Center, Prineville, OR

by

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Introduction

Riparian ecosystems are a high conservation priority because they provide a disproportionate quantity of ecosystem services relative to their extent on the landscape. In the arid western US, riparian ecosystems (those influenced by and adjacent to flowing water) and wetlands occupy from 0.8 to 2% of the landscape (NRC 2002, Naiman et al. 2005), yet they provide habitat, water, and other resources to greater than half the wildlife species in the region and harbor the highest plant, bird, insect, reptile-amphibian and mammal biodiversity of any terrestrial ecosystem. In addition, riparian areas subsidize aquatic ecosystems, provide linkages across and within landscapes for the passage of organisms and the exchange of material, perform important biochemical cycling and water quality functions, store groundwater, attenuate floods, serve as areas for agriculture, human development, and recreation, and are associated with a range of other services valued by humans and important ecologically (NRC 2002, Covich et al. 2004, Giller et al. 2004).

Human activities and extraction of natural resources on the landscape affect the physical processes that support many of the values provided by riparian ecosystems and alter the rate, quantity, and quality of these services (Table 1). There is a large and rapidly increasing human demand on resources associated with rivers throughout the western US. Activities such as road building, residential development, logging, ski area expansion, and mining influence the hydrologic cycle by affecting interception, timing and extent of snow accumulation and melt, infiltration, groundwater recharge, evapotranspiration, and runoff. Through influencing runoff, vegetation cover, and soils, such activities also affect the timing and rate of sediment yield from watersheds and the volume of sediment entering stream channels; key determinants of channel form and condition of riparian ecosystems. The ability of stream channels to transport or adjust to changes in sediment delivery is governed by the volume and timing of stream flow, which is affected by humans *indirectly* through the factors listed above and *directly* through water development (Poff et al. 2007, Schmidt and Wilcock 2008).

There is a need to quantitatively assess threats to ecosystem functions and to develop risk management strategies to assure that management resources are directed towards minimization of threats and maintenance of ecosystem function.

In conjunction with the uncertainties associated with climate change, an evaluation of the services, threats, trade-offs, and alternative approaches to continued utilization of riparian resources is timely (NRC 2004, Kremen and Ostfeld 2005). There is a need to quantitatively assess threats to ecosystem functions and to develop risk management strategies to assure that management resources are directed towards minimization of threats and maintenance of ecosystem function (Lowrance and Vellidis 1995).

Table 1. General listing of ecosystem services and goods associated with riparian ecosystems (Brauman et al. 2007).

Ecosystem Services	Ecosystem Goods
Erosion control	Areas for recreational, spiritual, and aesthetic opportunities
Mitigation of flood and droughts	Wildlife and fish habitat
Nutrient cycling	Water supply
Sediment, nutrient, and water transport	agriculture
Temperature regulation	commercial
Water and air purification	hydropower
Water storage and release	industrial
Wildlife habitat	Municipal, recreation, transportation

Goal of project

The goal of this report is to provide an initial, coarse-scale assessment of historical, current and future threats to streams and riparian areas in the western US. This effort is intended to support the development of a strategic vision for the future of western wildland management that offers strategies for managing these important landscape elements and their watersheds, recognizing the need to balance sometimes conflicting interests and demands. The mission of the Western Environmental Threat Assessment Center is early detection, identification, and assessment of multiple environmental threats "such as insect, disease, invasive species, fire, loss or degradation of forests, and weather-related risks" (Quigley et al. 2004). This report contributes to meeting this goal for one particular component of western landscapes--riparian ecosystems associated with flowing freshwater systems--and provides a framework for future assessment of condition and trends in these biophysically complex and temporally dynamic ecosystems. Our approach was to examine those factors that fundamentally influence riverine and riparian functioning through examining changes in the processes governing those functions; principally *flow regime*, *sedimentation*, and *lateral connectivity*. Each of these processes operates at different scales and is characterized by different aspects of the hydrologic system. Flow regime is conditioned by processes and patterns occurring throughout the entire watershed. Sediment delivery reflects upland conditions. Lateral connectivity measures human-modifications directly within the valley bottom and riparian areas (Reeves et al. 2006). Through characterizing these processes by modeling past, present, and projected future conditions, we examined the current status of streams relative to unaltered reference conditions and evaluated those riparian areas most at risk of future change under various future scenarios of climate change and human-caused land cover change. Note that because over 22 federal agencies, and many more state, local, and tribal agencies as well as private landowners and corporations manage the western landscape and its hydrologic cycle (Naiman et al. 2005), this assessment examines the entire landscape without regard for political boundaries.

The mission of the Western Environmental Threat Assessment Center is early detection, identification, and assessment of multiple environmental threats such as insect, disease, invasive species, fire, loss or degradation of forests, and weather-related risks.

Our approach was to examine those factors that fundamentally influence riverine and riparian functioning through examining changes in the processes governing those functions; principally flow regime, sedimentaiton and lateral connectivity.

Rivers and riparian ecosystems

Rivers are physically and biologically complex and dynamic ecosystems and may experience large seasonal and inter-annual variability. Rivers and riparian ecosystems are recognized as important areas for conservation as they provide a range of services to society, provide unique and productive habitat for wildlife, and serve as corridors – connecting otherwise disconnected landscapes through exchanges of water, sediment, nutrients, pollutants, and organic materials (Tockner and Ward 1999). The form of stream and river channels generally reflects the supply of sediment and the annual and interannual delivery of water reaching and passing through channels. The dynamics of water and sediment delivery is a function of upland vegetation cover, hillslope processes (angle, aspect, stability, and geology), groundwater patterns, climate, and land use activities throughout the watershed.

For systems in relative equilibrium or quasi-equilibrium, the hydrologic regime, slope, and channel dimensions maintain a net balance with supply of sediment delivered from upstream through the reach (Langbein and Leopold 1964). Such channels tend to maintain form (e.g., width, depth, shape, and slope) through time, adjusting and readjusting in response to rare floods or surges and depletions in sediment supply

from upstream. Planform and cross sectional form are maintained over time (fluctuating around a mean state) despite channel migration or channel changing events (Schumm 1977). Along meandering alluvial streams, erosion of cutbanks and deposition of sediment on point bars, drives the channel meandering process and results in chutes, meander cutoffs and oxbow lakes, and ridge and swale topography which together create a complex fluvial landscape that supports a diverse biota. Straight channels are the least dynamic of channel forms in the western US when compared to braided, island braided, and meandering systems, and have relatively slow floodplain turnover rates (Beechie et al. 2006). Nonetheless, occasional fluvial disturbance and moisture gradients maintain unique and species rich vegetation in mountain channels. Systems that have flashy, seasonally irregular, or extreme flow regimes with large fluctuations in sediment delivered to them may experience tremendous variation in channel form through time shifting between braided, anastomosing, and meandering. These non-equilibrium systems are common in arid and semi arid ecoregions in the western US (Friedman and Lee 2002, Merritt and Wohl 2003). Braided streams may have highly variable flow regimes (several orders of magnitude between low and high flows) and in some areas have the highest turnover of floodplain sediment and fluvial surfaces (Villarin et al. 2009).

Many of the same factors that contol the characteristics of riparian areas and support these ecosystems services when operating within a natural ('reference') range of variability may become stressors or threats to these services when they begin to operate outside of this range.

Many species rely upon fluvial processes for establishment and growth and possess adaptations to flow regime and fluvial disturbances associated with river meandering or occasional flooding (Karrenberg et al. 2002, Lytle and Poff 2004). Such processes support riparian vegetation and streambank condition which have been shown to be intimately linked with aquatic health and the health of anadromous fishes (Platts 1991). Further, occasional disturbance (including catastrophic flooding, drought stress, and disease) while stressing or killing individuals, may be beneficial to the ecosystem, enhancing fitness of extant communities and promoting decomposition, nutrient cycling, regeneration, recruitment and community heterogeneity (Kozlowski and Pallardy 2002). It is important to consider river dynamics and the range of natural variation in considering response of channels and adjacent riparian ecosystems to stressors.

Many of the same factors that control the characteristics of riparian areas and support these ecosystem services when operating within a natural ('reference') range of variability, may become stressors or threats to these services when they begin to operate outside of this range. The major determinants of stream channel form, processes, and ecological characteristics of riparian ecosystems include: 1) valley form, which constrains lateral channel movement, channel slope, and the influences of valley side slope processes on the channel, 2) sediment delivery to the channel, 3) seasonal and inter-annual hydrologic regime of the river and groundwater, and 4) connectivity of river channels throughout the river network and lateral connections between rivers and floodplains. Changes in valley form occur over geologic timescales ($>10^3$ years), with the exception of the valley-scale effects of dam construction. Changes in sediment delivery and hydrologic regime may be directly affected over relatively short timescales by humans and over longer timescales through factors associated with climate change. Changes in such processes may have different influences on different channel forms in different climates and valley settings.

Threats to riparian ecosystems

The threats to riparian ecosystems that have been identified in the literature include similar threats to those identified for upland ecosystems: invasive species, herbivory (both domestic livestock and wild ungulates), wildfire and fuels treatments, ecosystem fragmentation, drought, climate change, disease and insects, legacy impacts, urban development, mineral extraction, changes in hydrology, and geomorphic change (i.e., erosion and sedimentation; Table 2). In addition, rivers are particularly vulnerable to human water use and altered hydrologic regime (Graf 1999, Allan 2004, Dudgeon et al. 2006). Because channel processes and riparian areas are inherently tied to hydrologic and sediment regimes, activities and conditions from throughout

the watershed influence riparian conditions along a river segment or reach. Because of complex interactive and cumulative effects, lag times, stochasticity, and site-specific characteristics, predicting deterministic/cause and effect relations is difficult. However, general direction and magnitude of river change may be predicted given known changes in driving variables. In addition, generalization regarding change in systems in response to stressors is risky as different channels may respond to the same change in different ways. Common indicators of stress in ecosystems include reduced biodiversity, altered productivity, increased prevalence of disease, reduced efficiency of nutrient cycling, increased dominance of exotic species and smaller, short-lived opportunistic species (Naiman et al. 2005). In riparian ecosystems such indicators may be direct responses to reduced or altered flow regime that governs water availability and disturbance magnitude and frequency and changes in sediment regimes delivered from upstream hillslopes and channels.

Common indicators of stress in ecosystems include reduced biodiversity, altered productivity, increased prevalence of disease, reduced efficiency of nutrient cycling, increased dominance of exotic species, and smaller, short-lived opportunistic species.

Table 2. Threats to western riparian ecosystems.

Threat	Examples of causes	Examples of effects
*Changes in flow regime† and dewatering	Surface water: dams, diversions, changes in land-use, climate change; groundwater: pumping, land use change, climate change	Water stress of vegetation, shifts in plant species composition, homogenization of riparian zone, simplification of biota, isolation of floodplain from stream, changes in stream-riparian organic matter exchange and trophic dynamics, alteration of floodplain biogeochemistry terrestrialization, secondary effects (fragmentation, channel change)
*Channelization	Bank hardening, levee construction, structural changes in channel - - deepening, berm development, meander cutoff	Isolation of floodplain from stream, changes in fluvial processes, changes in hydraulics (aquatic habitat and channel forms), alteration of floodplain biogeochemistry
Invasive species	Introduction, altered processes in system that facilitate establishment & spread (e.g., herbivory, changes in flow regime)	Displacement of native species, formation of monoculture, changes in site characteristics (e.g., biogeochemistry, soil characteristics, changes in water balance), shifts in community composition, changes in habitat structure
Changes in sediment delivery to channel	ORV use, roads (drainage, gravel application), livestock/herbivore trampling, changes in vegetative cover in watershed and/or along channel, direct mechanical impacts to channel, dams, and diversions	Shifts in channel and floodplain form (through increased or decreased sediment delivery to channel), changes in channel processes, incision/aggradation
Herbivory	Domestic grazing, wild herbivores (predator control)	Bank trampling, compaction, vegetation changes (cover, composition), stream capture, nutrient inputs
Wildfire and fuels	Fuel buildup from invasive species, fire suppression, decadent vegetation, flood suppression, lack of flooding-slower decomposition of organic material	Increases in frequency and intensity of fires, loss of fire intolerant taxa, changes in the structure of riparian vegetation and habitat quality and distribution, subsequent shifts in biota

*equally significant; all others loosely ranked; †Magnitude, frequency, timing, duration, rate of change, and inter-annual variability in stream flow.

Methods

Our approach in this assessment of threats to riparian ecoystems was to utilize available geospatial data from the western US, known relationships and standard models of runoff and sediment yield, and past and future scenarios of climate and land-use change to characterize the landscape-scale processes influencing riverine and riparian areas. Because geospatial data needed to characterize fine-scale (grain) patterns and

processes (e.g., fire, invasive species, grazing, species composition, etc.) are inconsistent and incomplete for the western US, we chose to examine processes that are more readily measured using spatial datasets that occur at a broader-scale (grain), but that also have well understood, direct effects on those occurring at finer spatial scales. There is strong evidence that the basis of our assumption that changes in driving variables at a large landscape scale propagate through levels to smaller and smaller spatial scales, influencing form and biological patterns (Poff 1997, Walsh et al. 1998, Jensen et al. 2001, Fausch et al. 2002, Poole 2002, Allan 2004, Burcher et al. 2007). We summarized the potential ecological condition of riparian areas at a range of scales using water resource regions (Hydrologic Unit Code 2) and watersheds (Hydrologic Unit Code 8; Seaber et al. 1987).

There is strong evidence that the basis of our assumption that changes in driving variables at a large landscape scale propagate through levels to smaller and smaller spatial scales, influencing form and biological patterns.

Through integrating available geospatial data, existing approaches, and new innovative modeling, we were able to develop an ecological risk assessment for comparing past (reference), present, and future states of riparian ecosystems.

As with other studies of watershed condition, we initially generated maps of a variety of indicators that have been commonly used to evaluate watershed impacts (Moyle and Randall 1998, Tiner 2004, Scott 2006, Brown and Froemke 2007, Mattson and Angermeier 2007, Sowa et al. 2007), including the proportion of watersheds in urban and/or cropland land use and various measures associated with roads, including road density and number of road-stream crossings (Figure 1). In addition, we calculated a new metric reflecting the influence of roads on stream channels that provides a measure of road configuration within a watershed by weighting location of a road by the inverse overland-flow distance to the nearest stream (Figure 2).

Through integrating available geospatial data, existing approaches, and new innovative modeling, we were able to develop an ecological risk assessment for comparing past (reference), present, and future states of riparian ecosystems. Ecological risk assessment is a process for evaluating the probability of adverse ecological effects as a result of exposure to one or more stressors (RAF 1992). Terms associated with risk assessment (e.g., risk, sensitivity, resilience, threat, and vulnerability) have a variety of definitions and connotations within different disciplines. Risk is a function of the perturbation, stressor, or stress and the vulnerability of the exposed system or component (Blaikie et al. 1994). Resilience is defined as "a system's ability to bounce back to a reference state after a disturbance and the capacity of a system to maintain certain structures or functions despite disturbance" (Turner et al. 2003). Turner et al. (2003) define vulnerability as "…the degree to which a system, subsystem, or system component, is likely to experience harm due to exposure to a hazard, either a perturbation of a stress/stressor." It continues to be a challenge to quantify the amount of disturbance or stress (or potentially interacting stressors) that an ecosystem can tolerate before it shifts to a different or degraded state, but understanding the processes that govern functioning systems can help us to better understand the magnitude and direction of change in the characteristics of a system in response to changes in these processes.

We utilized existing geospatial datasets and climate change scenarios available at broad spatial scales to model three factors to capture the primary processes that operate at smaller spatial scales to influence water and sediment yield from landscapes to and through stream channels: 1) longitudinal flow regime; 2) upland processes and sediment production; and 3) lateral connectivity and land use modifications in the riparian zone.

We have taken a process-based approach – that is, we assess riparian areas in the West by quantifying the magnitude and direction of change of the primary ecological factors that control pattern and process in riparian ecosystems. Many of the factors considered threats to natural ecosystems by land mangers (such as insects, pathogens/disease, invasive species, fire, loss or degradation of forests, weather-related risks, and other episodic events; Quigley et al. 2004) are difficult or impossible to quantify at regional scales as available geospatial data are discontinuous, inconsistent, or unavailable. However, many of these 'threats' are conditioned by quantifiable factors operating at broader spatial extents. Therefore we utilize existing geospatial datasets and climate change scenarios available at broad spatial scales to model three factors to capture the primary processes that operate at smaller spatial scales to influence water and sediment yield from landscapes to and through stream channels: 1) longitudinal flow regime; 2) upland processes and sediment production; and 3) lateral connectivity and land use modifications in the riparian zone.

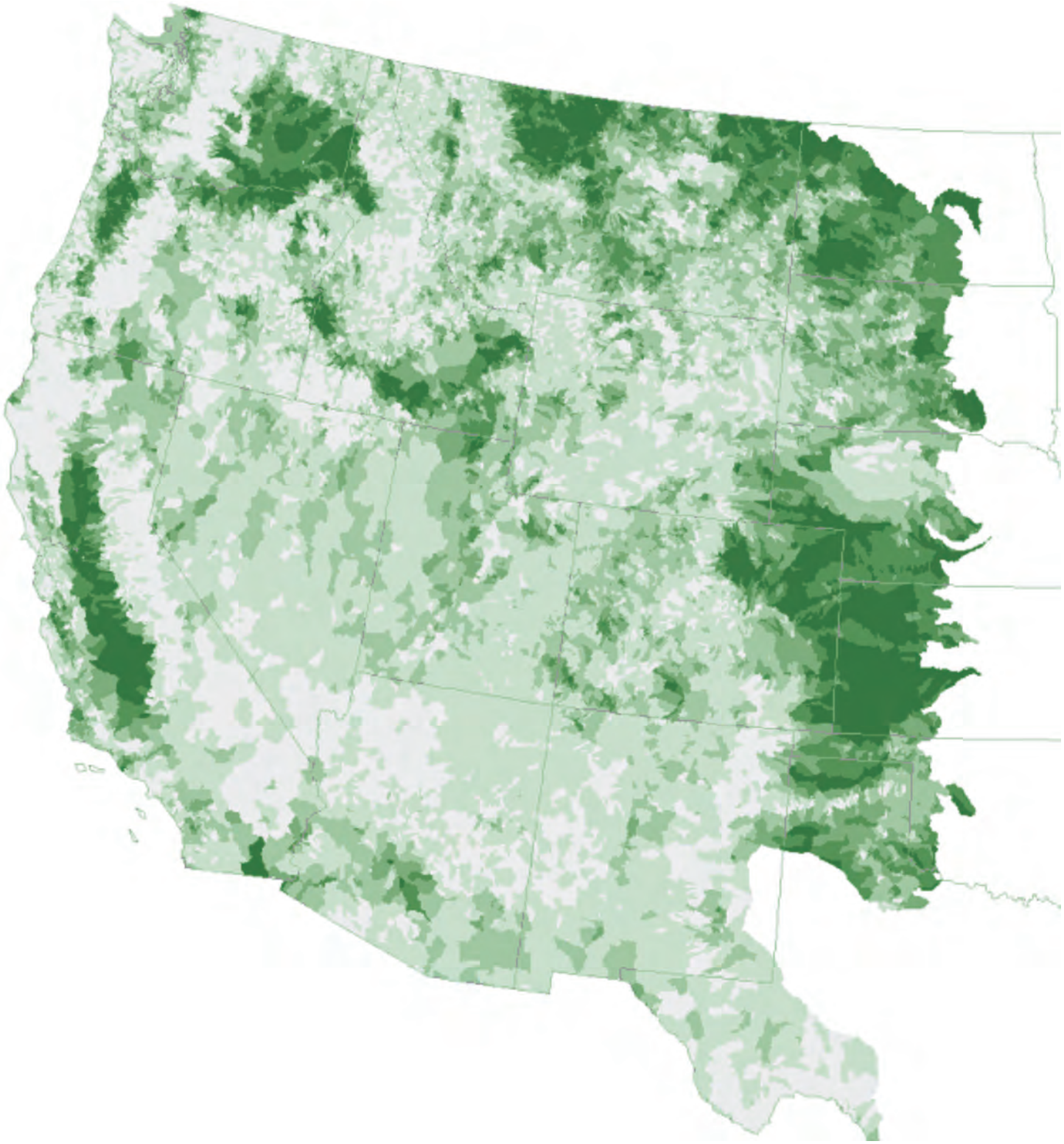

Figure 1(a). Dark green shows higher proportion of the average percent cropland in 2001 in each catchment.

Figure 1(b). Dark green shows proportion of cropland accumulated downstream through the catchments.

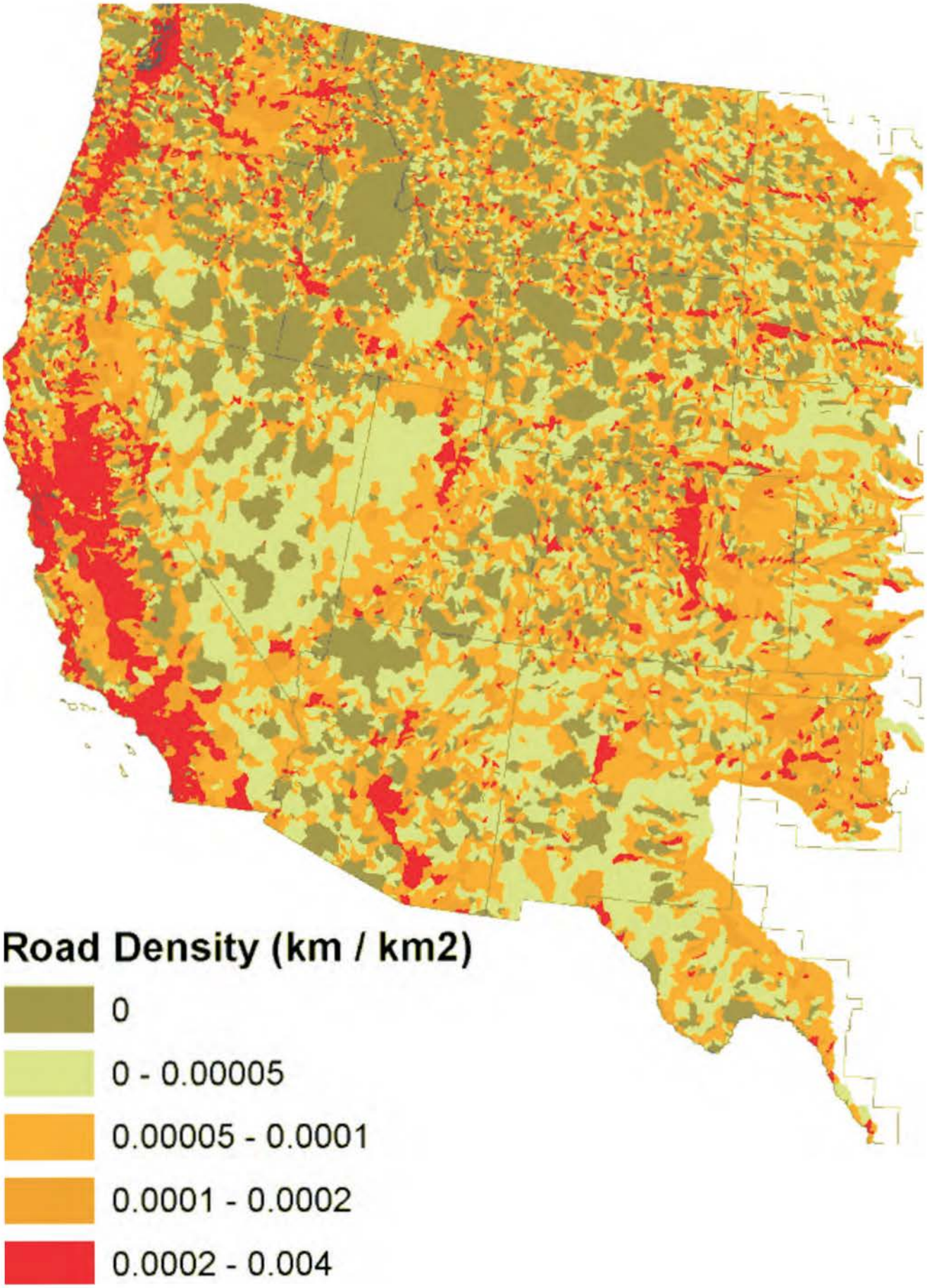

Figure 2(a). Catchments with higher average road density (km/km2) are shown in red.

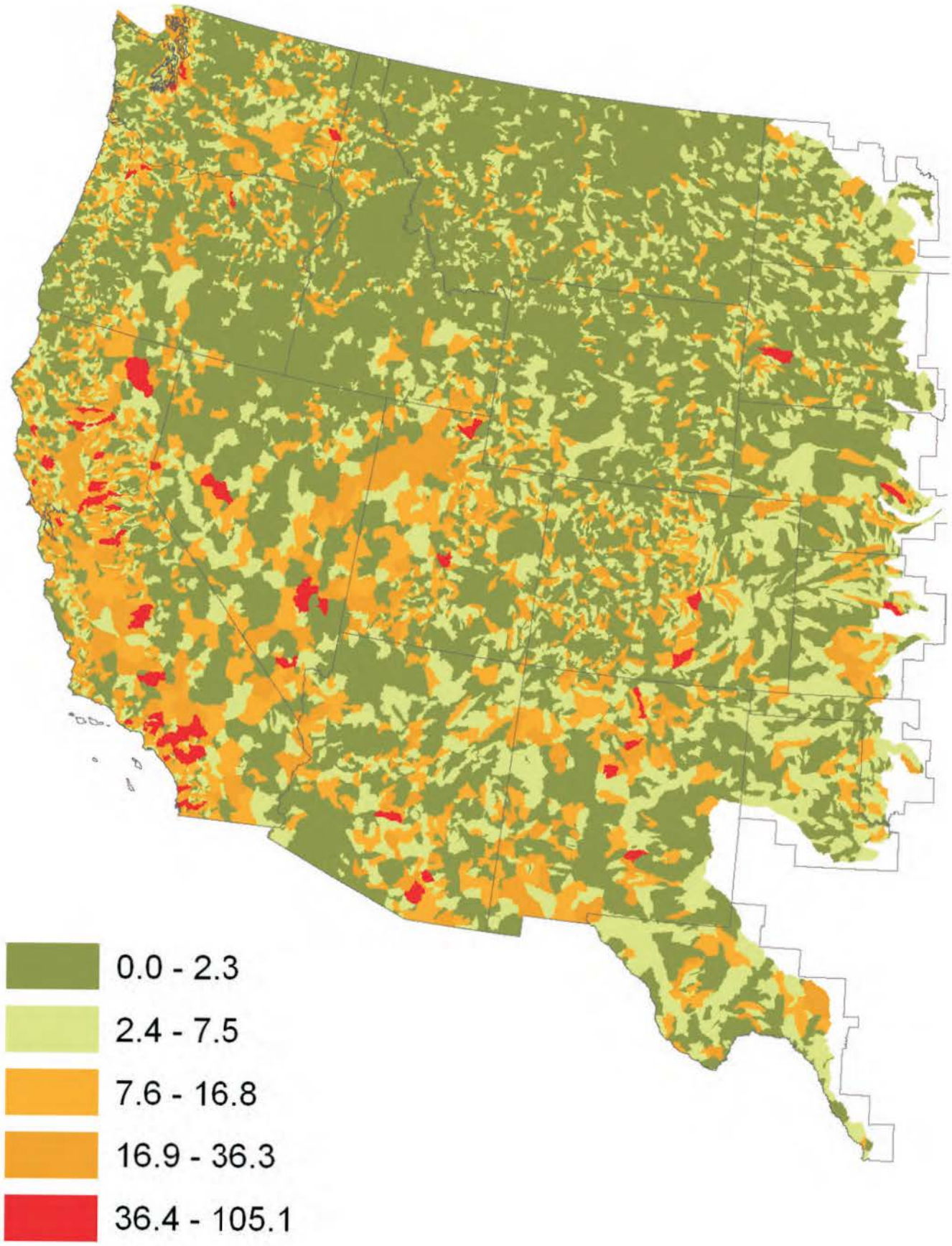

Figure 2(b). Map shows the average road density weighted by inverse distance to streams by catchment.

We used the USGS National Hydrography Dataset (NHD; 1:100,000 scale) to identify stream reaches, and 30 m resolution elevation and other data where available. We grouped streams from the NHD that were represented by the same unique reach from a coarser dataset, the US EPA's Reach File 1.2 (Hall et al. 2000). This resulted in reach catchments that are roughly comparable to HUC 6 to 7 level/(12 and 14"). To organize our assessment, we developed three scenarios to characterize conditions in the past (1900-1940), present (1940-2000), and future (2000-2030). We selected our time period to break at 1940 to be consistent with other studies that have shown that atmospheric CO_2 began to increase rapidly around 1940 (Soon et al. 1999; Table 3). This is also roughly consistent with the boom in urbanization and sprawl associated with the post-second World War development and construction of the interstate highway system.

Our reasoning in using these scenarios is to evaluate how much a given stream reach has deviated from antecedent conditions (e.g., past is roughly a "reference condition" for current conditions). Further, we distinguished two future scenarios to understand the relative contribution of two key threats: 1) urbanization and development and 2) changes in climate (primarily precipitation).

Table 3. For each of the three central processes that dominate riparian ecosystems, we generated three scenarios that reflect conditions in the past (1900-1940), current (1940-2000), and future (2000-2030).

SCALE	LONGITUDINAL Flow modification of natural discharge	UPLAND Sediment production (RUSLE)	RIPARIAN ZONE Lateral connectivity: % human-modified land cover in potential riparian zone
Past	Historical (1900-1940) mean annual precipitation and temperature, no dams	Biophysical setting (potential natural vegetation)	Biophysical setting (potential natural vegetation)
Present	Current (1940-2000) mean annual precipitation and temperature, with dams in 2004	Existing land cover (2005), housing density (2000), and roads (2003)	Existing land cover (2005), housing density (2000), and roads (2003)
Future	Projected (2000-2030 for A1B and B1) mean annual precipitation and temperature, 2007 dams	Existing land cover (2005), housing density (2030), and roads (2003)	Existing land cover (2005), housing density (2030), and roads (2003)

We estimated changes in the stream flow regime using a measure *F* we call "flow modification", which is measured as the ratio of the normal storage volume of a dam (acre feet) to the natural mean annual "virgin" discharge (assuming no dams or other human modifications). We summed the normal storage volume of dams cumulatively downstream, using data from the National Inventory of Dams (NID; USACE 2008). Our measure of characterizing flow modification is consistent with the literature (see Graf 1999; Dynesius and Nilsson 1994, Nilsson et al. 2005). Values of $F = 0.0$ indicates no flow modification; a value of $F = 1.0$ indicates that reservoirs are able to store roughly the average annual discharge flowing through a given stream segment.

We used the Revised Universal Soil Loss Equation (RUSLE; Renard et al. 1997) to estimate the amount of sediment production. RUSLE uses five major factors to compute the average annual erosion or sediment produced in a watershed: rainfall erosivity (R), soil type (K), topography composed of length (L) and slope (S),

cover type (C), and management practices (P). To compute RUSLE, we generated a series of raster datasets at 30 m resolution for each of the RUSLE factors.

To estimate the direct loss of riparian zone areas, as well as the loss of lateral connectivity, we measured the proportion of human-dominated land uses inside the valley bottom. Human-dominated land cover types within the riparian zone were estimated based on reclassified urban and agricultural cover types from LANDFIRE to be 1.0 and all other "natural" types 0.0. We also included other areas in the riparian zone that had either major roads (highways, secondary roads) or at least exurban (1 unit per 16 ha) or higher housing density.

Table 4 provides a summary of the three process factors and datasets used. We also standardized the raw data from the individual factors to compute a composite index we call the "riparian threats score" that could be used to evaluate the relative "most" from the "least" affected watersheds. To do this for flow fragmentation, we standardized the current (2006) raw values using the mean and standard deviation computed from the historic (1940) values. Similarly, we standardized the flow fragmentation for future scenarios using the mean and standard deviation of the current (2006) values. This standardization resulted in values that range (roughly) from -1 to +1, where negative values denote a reduction in flow fragmentation and positive values denote an increase in fragmentation. Please see Appendix 1 for more detailed methods.

Table 4. Summary of the datasets used to generate the scenarios in our analyses.

SCALE	LONGITUDINAL Flow fragmentation: modification of natural discharge	UPLAND Sedimentation: sediment production (RUSLE)	RIPARIAN ZONE Lateral connectivity: Proportion of human-modified land cover in riparian zone
Past	PRISM 1900-1940 (mean annual, +/- 1SD)	LANDFIRE Biophysical Settings	LandFire BpS
Present	PRISM 1940-2000 (mean annual +/- 1 SD)	LANDFIRE Existing Vegetation Types StreetMap 2006 roads* SERGoM 2000	LANDFIRE Existing Vegetation Types StreetMap 2006 roads
Future	Climate change Precip/temperature 2030	Land use change LandFire EVT StreetMap 2006 roads SERGoM (2030 and change 2000 – 2030)	Land use change LandFire EVT StreetMap 2006 roads SERGoM (2030 and change 2000 – 2030) Impervious surface

* Environmental Systems Research Institute (ESRI) (2007)

Results

1. Longitudinal connectivity and flow fragmentation

The average flow modification *F* indicates that reservoirs can store between 16% to 200+% of the annual stream discharge (Table 5), with lower modification for the regions of the Pacific Northwest (~18%) and the upper Missouri (~47%), and higher modification in the Rio Grande (~272%), Great Basin (~225%), and Colorado (~220%) basins. However, flow modification is highly variable spatially (Figure 3), with about 10% of catchments having at least 100% modification, 16% having 50% modification, and 23% have at least 20% modification. About 55% of catchments having 0% flow modification, meaning no major reservoirs located within or above a catchment.

Watersheds most highly modified occur mostly in lower Colorado R. and Great Basin Regions.

Our measure of flow modification changes from the past to current scenarios as a function of climate differences between our past (1900-1940) and current (1940-2000) time periods – we assumed the dams and storage volumes do not change over our scenarios to isolate changes as a function of climate changes. Flow modification will likely decrease (current to future, both A1B and B1 climate change scenarios) for the Arkansas and Rio Grande regions (because of increased discharge), but will likely increase substantially for the Colorado, California, and to a lesser extent the Great Basin regions. We also provide a map that depicts the "most" and "least" effected of HUC8 watersheds - across the West - based on ranks of the flow modification values averaged by HUC8 (Figure 4). Best condition watersheds appear mostly in the Pacific Northwest headwaters, but also a few notable watersheds in the Arizona and New Mexico. Watersheds most highly modified occur mostly in the lower Colorado (15) and Great Basin regions. We normalized the values by region (i.e. ranked the raw values within a region, and then displayed the distribution of values by classifying the values by 5% increments). Normalizing by region provides a more contextualized ranking of each watershed, highlighting, for example, least effected condition watersheds in most of the watersheds, while many of the lower elevation HUCs are more affected. Notably the Big Thompson and St. Vrain watersheds just north of Denver on the Colorado Front Range are in the top 5% of region 10 in terms of flow modification.

Table 5. The percent of flow modification in the past (1900-1940), current (1940-2000), and future (2000-2030) for water resource regions (HUC2).

			Flow modification			
Water Resource Region (WRR)	Number of dams[1]	Total storage (M acft)	F_{past} (1900-40)	F_{curr} (1940-00)	F_{a1b} (2000-30)	F_{b1} (2000-30)
Arkansas* (11)	526	2.5	0.97	0.90	0.62	0.61
California (18)	1,536	70.6	0.72	0.65	0.85	0.84
Colorado (14-15)	1,630	76.4	2.17	2.20	2.43	2.47
Great Basin (16)	888	7.1	2.25	2.19	2.21	2.22
Missouri* (10)	8,287	79.0	0.47	0.44	0.41	0.41
Pacific NW (17)	2,147	66.1	0.18	0.16	0.15	0.15
Rio Grande (13)	398	7.3	2.72	2.74	2.65	2.66

* denotes regions with only partial occurrence in the western US study area.

[1] Using National Inventory of Dams (2007)

Figure 3. The spatial variability of flow modification (%). Not surprisingly, the higher order streams have the most severe and widespread flow modification while lower order streams are the least flow regulated.

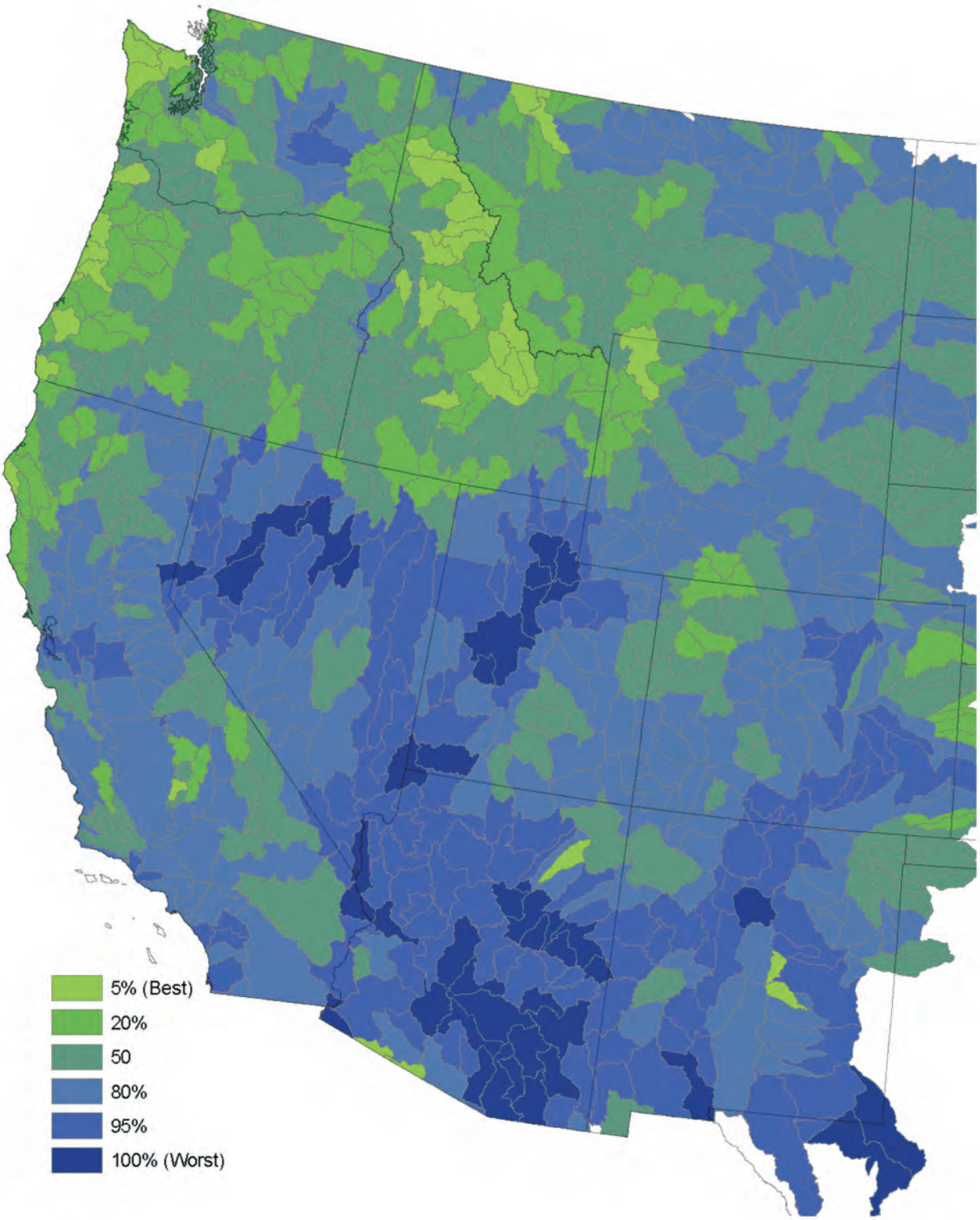

Figure 4(a). Flow fragmentation for the current period summarized by HUC8 watersheds, the least (bright green) and the most (dark blue) affected watersheds. Note that the upper and lower Colorado basins (WRR 14 & 15) is normalized as one region.

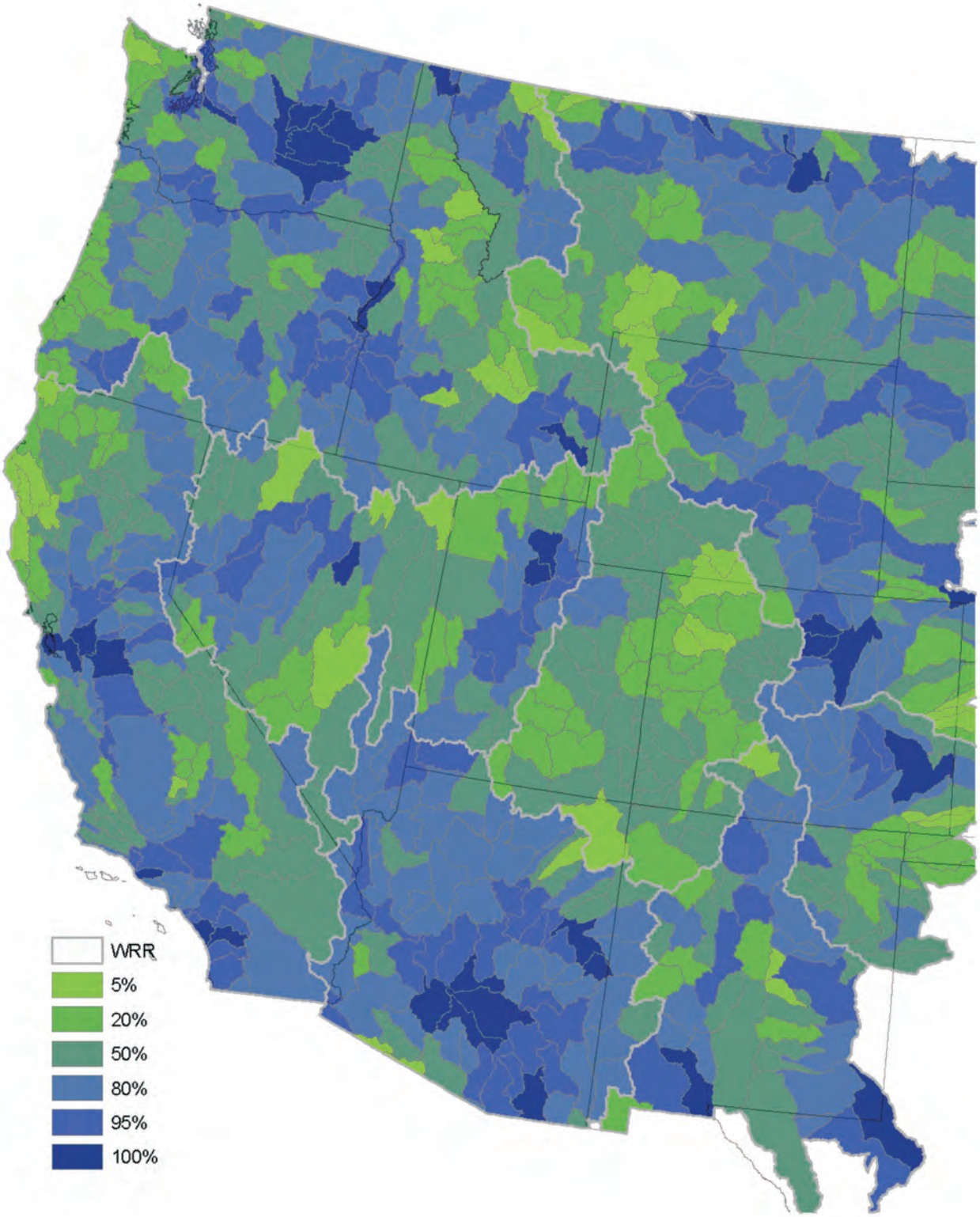

Figure 4(b). Flow fragmentation for the current period summarized by HUC8 watersheds, the least (bright green) and the most (dark blue) affected watersheds. This map shows flow fragmentation normalized by water resource region. Note that the upper and lower Colorado basins (WRR 14 & 15) is normalized as one region.

Major factors causing impacts

The ecological consequences of reduced streamflow associated with water storage, extraction, or altered flow regimes in stream include reduced channel capacity and aquatic habitat, encroachment of upland vegetation into riparian areas (terrestrialization), reduced extent of riparian zones, and drought stress-related shifts in the distributions of plant and animal populations and communities. Altered flow regimes may cause changes in riparian plant species richness (Nilsson et al. 1991, Jansson et al. 2000), plant growth and productivity (Stromberg and Patten 1990), community composition (Merritt and Cooper 2000, Merritt and Wohl 2006) and loss of riparian forests (Rood and Mahoney 1990, Braatne et al. 2007). Susceptibility of vegetation to disease, insect infestation, drought-related reductions in cover and health, and susceptibility to fire-related mortality are likely to increase in riparian areas as a result of increased water stress associated with altered stream flow. In addition to direct effects of altered flow regime, flow and sediment related changes in channel form and channel processes affect the characteristics and functioning of riparian areas as well.

In addition to disadvantaging native, flow-adapted species and advantaging ruderal, generalist and non native plants, altered flow regime often disrupts connectivity along rivers, altering rates and quanities of nutrient, sediment, and organic material transport and the movement of propagules and organisms.

Because riparian biota are often adapted to the timing of components of stream flow regime (e.g., peak flows, low flows, etc.), shifts in the timing of such flows will likely decouple specific life-history stages of plants from environmental cues and suitable abiotic conditions (Lytle and Poff 2004, Merritt et al. 2010). Such decoupling can have important consequences for germination, establishment and growth of riparian plants. For example, riparian cottonwood and willow (members of the Salicaceae family) are adapted to disperse seed in synchrony with the falling limb of the snowmelt hydrograph when habitat is most available and best suited to seedling establishment (Stella et al. 2006). Cottonwood forest collapse downstream from dams is widespread in western North America and has been attributed to flow alteration associated with dam operations (Rood and Mahoney 1990). Structured population models of altered streamflow regime associated with river regulation and/or climate change project declines in cottonwood populations and cottonwood-dominated riparian ecosystems for centuries into the future as a result of reduced high and low stream flow and altered timing of flow (Lytle and Merritt 2004). Regional studies in the southwest (Stromberg et al. 2007) and throughout the American west (Merritt and Poff 2010) suggest that decreases in streamflow can result in shifts in riparian vegetation from native-dominated (*Populus* spp.) riparian forests to non-native dominated (*Tamarix* spp.) riparian shrublands. Such shifts have implications for habitat quality, structural complexity, and thermal regimes along flow regulated rivers. In addition to disadvantaging native, flow-adapted species and advantaging ruderal, generalist and non native plants, altered flow regime often disrupts connectivity along rivers, altering rates and quantities of nutrient, sediment, and organic material transport and the movement of propagules, and organisms (Nilsson et al. 2005).

Decreases in high flows in streams is likely to result in decreased lateral connectivity between streams and their floodplains due to decreases in the frequency, magnitude and duration of overbank flows. Decreases in overbank flows can have detrimental effects on riparian ecosystems, leading to decreased productivity through altering rates of decomposition and nutrient cycling (Molles et al. 1998). Decreases in nutrient availability can lead to declines in the health and growth rates of riparian species and changes in species composition (Harner and Stanford 2003). Flow depletion resulting in longitudinal fragmentation of streams can also lead to dramatic shifts in the composition and structure of riparian areas (Stromberg et al. 2007). Formerly tree-dominated areas can experience shifts from forest to shrub-dominated systems along arid-land streams (Stromberg et al. 2007, Merritt and Poff 2010), resulting in reductions in wildlife habitat quality and complexity.

Flow related moisture stress has been associated not only with direct effects on plants, such as wilting, branch dieback, and death, but has also been linked to increased vulnerability of riparian trees to maladies associated with insects and disease (Maxwell et al. 1997). Generally, altered stream flow regimes may lead to reduced fitness of flow-adapted species, shifts in species composition in riparian areas, shifts in the functioning of communities, and changes in the quantity and quality of important ecological services associated with riparian systems (Nilsson and Svedmark 2002). This includes the volume of freshwater delivered for downstream use.

Climate change implications

Projections of climate change in the western US foretell of "water shortages, lack of storage capability to meet seasonally changing river flow, transfers of water from agriculture to urban uses, and other critical impacts" (Barnett et al. 2008). Changing climate in the western US is expected to affect streamflow in rivers and streams in a number of ecologically significant ways. Warmer temperatures may affect the timing and magnitude of stream flow through changes in precipitation, and temperature-related changes in evapotranspiration and snow and glacier accumulation (mass-balance) as well as timing and rate of melt (Tague and Grant 2009). Climate change is also expected to result in riparian vegetation shifts in response to the direct effects of rising temperatures and changes in seasonal and spatial distributions of moisture independent of stream flow (e.g., directly from atmosphere and non alluvial groundwater; Merritt et al. 2010).

Over the past half-century, climate change has caused increases in mean and extreme annual temperatures, changes in the spatial distribution and form of precipitation (e.g., snow versus rain), and the timing of runoff (Barnett et al. 2008). Estimates of future warming rates for the West are in the range of 2°–5°C over the next century (Cubasch et al. 2001). Trends in increasing winter precipitation falling as rain instead of snow in mountainous parts of the western US are widespread and are expected to continue (Knowles et al. 2006). In addition, climate change is expected to influence mean annual runoff, and stream flow timing and variability (e.g., interannual coefficient of variation; Barnett et al. 2008, Luce et al. 2009). Trends toward spring snowmelt beginning and ending earlier in the spring are expected to affect the timing and magnitude of peak flow in streams and rivers throughout the West through causing earlier and lower magnitude peak flows (Mote et al. 2005). Furthermore, summer low flows are expected to decrease in rivers throughout the West (Dettinger et al. 1995, Cayan et al. 2001, Luce et al. 2009).

Currently, more than 70% of stream flow in the western US originates as melting snowpack in the mountains (Hamlet et al. 2005). Mountainous areas of the western US where winter temperatures currently approach 0°C are most vulnerable to even modest climate warming (Hamlet et al. 2005). Coastal mountains in northern California and the Pacific Northwest will presumably experience the first shifts from snowmelt driven hydrographs to those responding to rain. Such streams are likely to experience trends towards winter-dominated runoff and lower summer flow volumes (Mote et al. 2005). Though these regions currently exhibit the lowest dam-caused fragmentation, reductions in snowpack and total annual runoff, could cause the existing infrastructure to result in higher *F* in coming years. Further, these regions are experiencing a surge in micro-hydropower operations that will also influence instream flows and riparian vegetation.

The same basic principles used to relate riparian vegetation change to stream flow attributes apply regardless of the cause of stream flow change (e.g., climate change-related flow alteration or dams, diversions, etc.; Davis et al. 2005, Merritt and Poff 2010). Documented responses of riparian vegetation to specific human-caused flow alteration scenarios can provide insight into changes that might occur in response to projected changes due to climate change. For example, groundwater depletion caused by pumping may result in dramatic shifts in aridland riparian communities such as conversion from riparian gallery forest to shrubland (Stromberg et al. 1996). Similarly, altered flow regimes may result in dominance shifts in riparian species (Stromberg

Through simultaneously affecting stream flow, extra-river-related moisture sources, and atmospheric conditions (e.g. humidity and temperature), climate change has the potential to have even more severe effects on riparian ecosystems compared to the direct effects of flow alteration.

et al. 2007, Merritt and Poff 2010) and significant changes in age-class structure of riparian forests (Lytle and Merritt 2004). Through simultaneously affecting stream flow, extra-river-related moisture sources, and atmospheric conditions (e.g., humidity and temperature), climate change has the potential to have even more severe effects on riparian ecosystems compared to the direct effects of flow alteration. In conjunction with flow alteration and increased water extraction from streams, rivers, and groundwater, climate change has the potential to constitute a significant threat to riparian ecosystems. However, climate change will influence rivers in different ecoregions and valley settings differently; at the regional scale, this may result in redistributions of different types of ecosystems on the landscape. For example, if the transition between arid and semi arid regions shifts northward, it is possible that those arid land riparian ecosystems could migrate north over some period of time. The factors that interact with climate change and human-caused stream flow alteration may buffer against change or accelerate it.

In conjunction with flow alteration and increased water extraction from streams, rivers, and groundwater, climate change has the potential to constitute a significant threat to riparian ecosystems.

Interacting factors

The combined effects of increased human demand for freshwater resources, increased water development, and decreased in water supply due to climate change are likely to put intense and widespread stresses on riparian ecosystems. Since these systems currently occupy such a small percentage of the entire western landscape, the relative importance of riparian corridors for wildlife habitat and migration will increase with a warmer, drier climate. Combined with rising temperatures, reduced stream flow is likely to place additional stresses on stream flow and groundwater dependent plants and favor drought-tolerant and generalist species. In some cases, the consequences of climate change may be suppressed by dam operations in cases where baseflows are maintained despite drought cycles. However, water storage and extraction are likely to have negative effects on flow-adapted riparian species.

Whereas flow regimes in western states historically were regionally unique and supported ecosystems, flow management has resulted in rivers that are currently more similar to one another across regions.

Disturbed sites are generally more invasible than sites with well-established vegetation. Any form of disturbance that alters vegetation cover, kills plants or removes biomass can affect space and resources of a site and influence its susceptibility to invasion. Removal of biomass through climate change-related mortality, mechanical means, grazing, and or fire is strongly associated with invasibility of sites by ruderal or non-native species. If the species or communities replacing former, flow-adapted communities are more productive or have a different growth form, they may cause differences in habitat qualities and functioning of these systems.

Increased mortality of riparian plants combined with decreased flooding and decomposition on floodplains could interact to influence fire intensity and fire-related mortality in riparian areas. Increasing fire intensity can lead to fire-related mortality and further shifts from riparian species to those adapted to fire. Such shifts have occurred along the Middle Rio Grande in New Mexico, where fuel accumulation on floodplains has facilitated high intensity fires that can result in replacement of native riparian forest by fire-tolerant shrub species (Figure 5).

The combined effects of climate change, increased human demands for water, and continued water development is a major threat to native riparian habitats throughout the West (Baron et al. 2002). Whereas, flow regimes in western rivers historically were regionally unique and supported a diversity of ecosystems, flow management has resulted in rivers that are currently more similar to one another across regions (Poff et al. 2007). Combined with increased susceptibility to invasion by non native plants, shifts in community dominace, altered fire regimes, and other factors related to water redistribution and scarcity, human and climate-caused

flow changes may be the most widespread threat to rivers in the western US. However, the water infrastructure (e.g., dams, reservoirs, diversions, canals, and pipelines) provides opportunities for strategically managing flows to maintain desirable attributes of some systems while still meeting society's demands for water (Poff et al. 2010).

*Figure 5. Shift in dominance from native cottonwood (*Populus deltoides*; dead overstory) forest to non-native salt cedar* (Tamarix ramosissima*; monotypic understory) shrubland following intense wildfire on middle Rio Grande, NM.* Tamarix *contributed significantly to fuel loads and ladder fuels in this system. Photograph by D. Merritt.*

Changes caused to delivery of ecosystem services

Reduced extent of riparian areas results in direct and easily quantifiable losses in such ecosystems services as wildlife habitat and recreation. Less quantifiable are the loss of aesthetic values associated with reduced riparian cover and changes in species composition and physiognomy.

Any reduction in riparian extent on a landscape results in direct losses of the quantity and quality of the ecosystem services provided by riparian areas. Reduced extent of riparian areas results in direct and easily quantifiable losses in such ecosystem services as wildlife habitat and recreation. Less quantifiable are the loss of aesthetic values associated with reduced riparian cover and changes in species composition and physiognomy. Riparian areas contribute to higher water quality in streams through trapping sediment and pollutants from upslope areas and reducing the volume introduced to stream channels (Johnson and Buffler 2008). Reduction in the width of riparian areas associated with reduced stream flow volume and

lower peak flows in the western US, will result in lower buffering capacity between aquatic and upland habitats. Flow-related reductions in species composition and cover of riparian vegetation can have cascading effects on stream channel morphology and water quality. Reduced vegetative cover may result in compromised bank stability, less infiltration of runoff, increased erosion, and increased sediment entering stream channels. Reduced flows can reduce the competence of a stream to transport sediment entering channels, resulting in further channel change.

Reduced riparian width and vegetative cover make stream channels and aquatic ecosystems more susceptible to degradation associated with activities in riparian areas. Within riparian areas, recreation, livestock grazing, browsing, and burrowing wildlife, and other activities that disturb vegetation or soil all influence riparian characteristics. Less extensive and more degraded riparian areas are more vulnerable to degradation from peripheral activities such as logging, road construction and use, upland fire, discharge of pollutants, agricultural activities (nutrients, herbicides, pesticides and sediment), and other activities that may have an effect on valley bottoms. Less degraded and more extensive riparian areas are more resilient to perturbations and the more resistant they are to change due to such stressors.

However, protecting instream flows for ecosystems and their services ("environmental flows") at the expense of human demands for water will become increasingly contentious with future population growth, climate change, and continued water development throughout the West.

Management implications

Awareness of the negative influences of altered flow regimes on streams, rivers and riparian areas has led to the development of methods of quantifying the effects of such change and managing flows to minimize negative effects on aquatic and riparian ecosystems (Poff et al. 2007, Dudgeon et al. 2006). The most significant influence on river flow regimes in the western US are dams, water diversions, and groundwater pumping. Mechanisms for protecting streams from water depletion and associated degradation to riparian areas include Federal laws (e.g., the Clean Water Act and the Endangered Species Act), instream flow programs administrated by state agencies, licensing of hydropower facilities through the Federal Energy Regulatory Commission (FERC), and land management plans specific to public land management agencies.

Methods for strategically designing flows so that aquatic and riparian biota are accommodated without compromising human water needs are beginning to be more widely developed and applied (Poff et al. 2010). However, protecting instream flows for ecosystems and their services ("environmental flows") at the expense of human demands for water will become increasingly contentious with future projected population growth, climate change, and continued water development throughout the West (Poff et al. 2003).

2. Upland processes and sedimentation

Changes in sedimentation from the past to current scenarios show small decreases in urban areas such as the Front Range of Colorado and Puget Sound, but also some localized watersheds throughout the West (Figure 6). Most of the changes, however, include large increases (>100%) in sediment produced watersheds. This occurs mostly in the eastern Washington area, Great Basin, central valley of California, and southern New Mexico. There are fairly subtle changes likely in the future (Figure 6) in terms of sediment averaged by watershed (HUC8). Some areas are likely to decrease in sediment due to land use changes that are dominated by urbanization (e.g., St. Vrain watershed on the Colorado Front Range, northern Idaho/western Montana, Willamette Valley). Few watersheds appear to have increased amounts of sediment in the future – near Boise,

Idaho and in north-central Arizona, for example. Changes in sediment estimated by our model are based only upon land use changes; potential changes in sedimentation due to climate change (e.g., change in precipitation intensity) are not modeled explicitly. A decrease in sediment can occur from urbanization – land use and cover changes to suburban and urban densities that have more impervious surface, and therefore decreased sedimentation (e.g., most of the Front Range of Colorado). Increases in sediment can occur because of rural to exurban land use changes. It is important to note that our model may underestimate sedimentation that can be caused by development, such as construction of houses and roads. This underestimation is likely due to the fact that we are measuring development at a particular time step (a "snap-shot"), rather than summing the incremental changes through time. That is, land use change from rural to urban development likely results in a near-term increase in sediment due to construction of homes and roads, but we only estimate sedimentation for urban land cover, which has low sediment yield because of high impermeability. Moreover, the RUSLE cover factors for exurban land uses are very approximate, and future research should provide more refined values for C (the cover management factor). Finally, recall that the future scenario does not include likely changes to roads (e.g., new and widened roads).

The watersheds most affected due to sedimentation changes are distributed throughout the West, with no strong spatial pattern (Figure 7). The watersheds in worst condition appear to cluster the Great Basin region, in south-central CA, central OR and eastern WA, and southern NM. Normalizing by water resource region enables more regional evaluation.

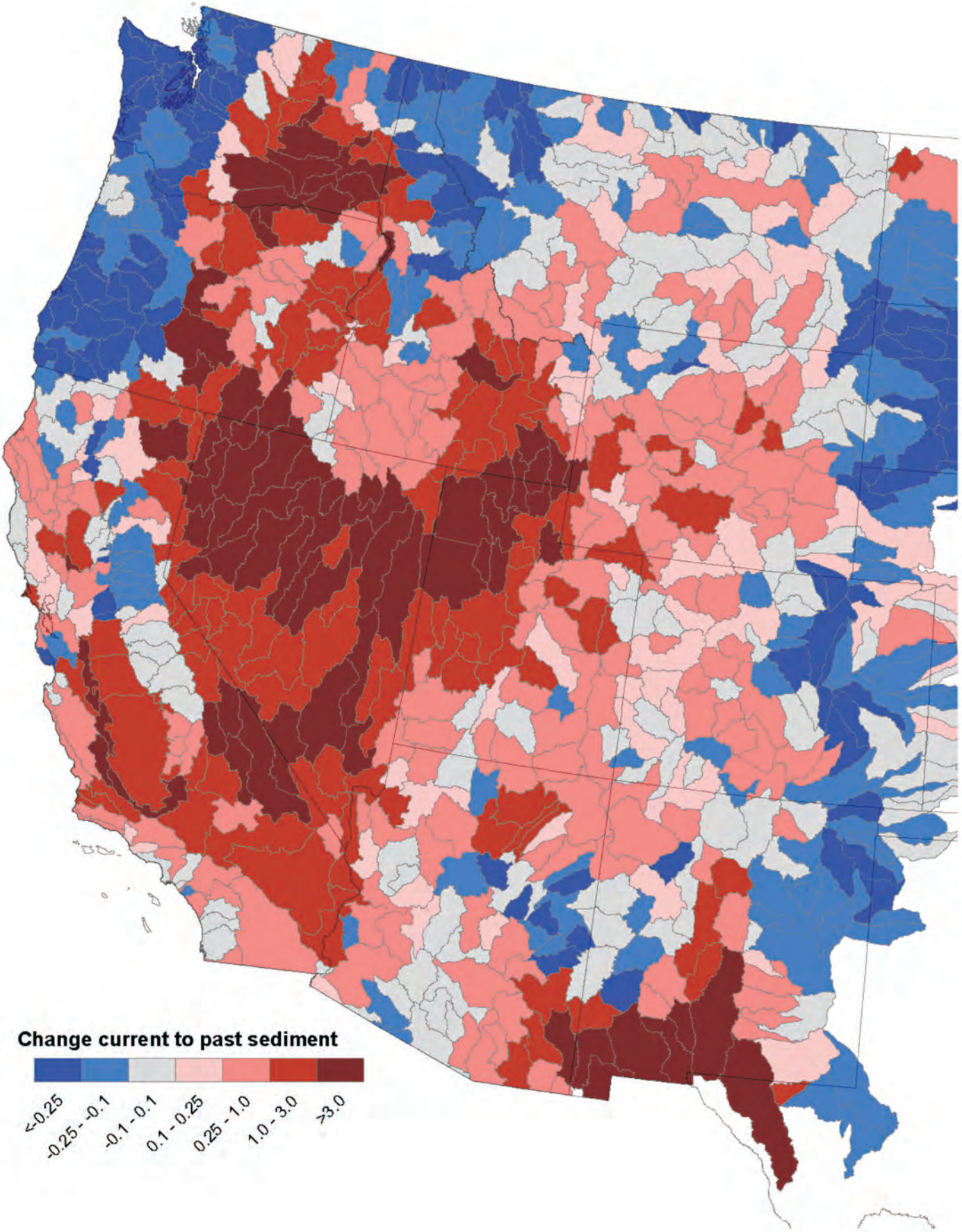

Figure 6(a). Changes in sedimentation from the past to current scenario, averaged by HUC8 watershed. Decreases (shown in dark blue) in sediment can occur in our model due to urbanization, which results in more impervious surface, and therefore, decreased sedimentation.

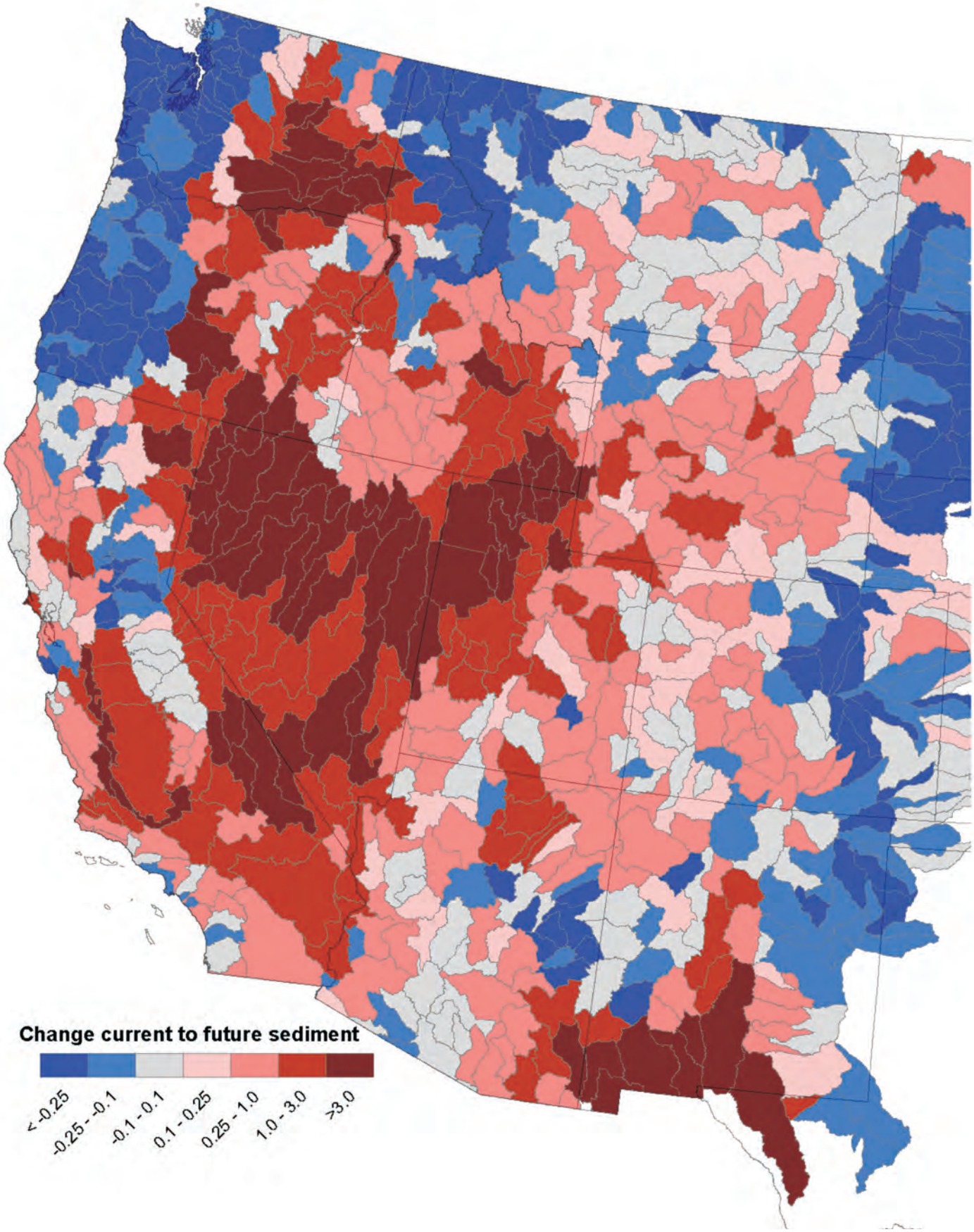

Figure 6(b). Changes in sedimentation from the current to future scenario, averaged by HUC8 watershed. Decreases (shown in dark blue) in sediment can occur in our model due to urbanization, which results in more impervious surface, and therefore, decreased sedimentation.

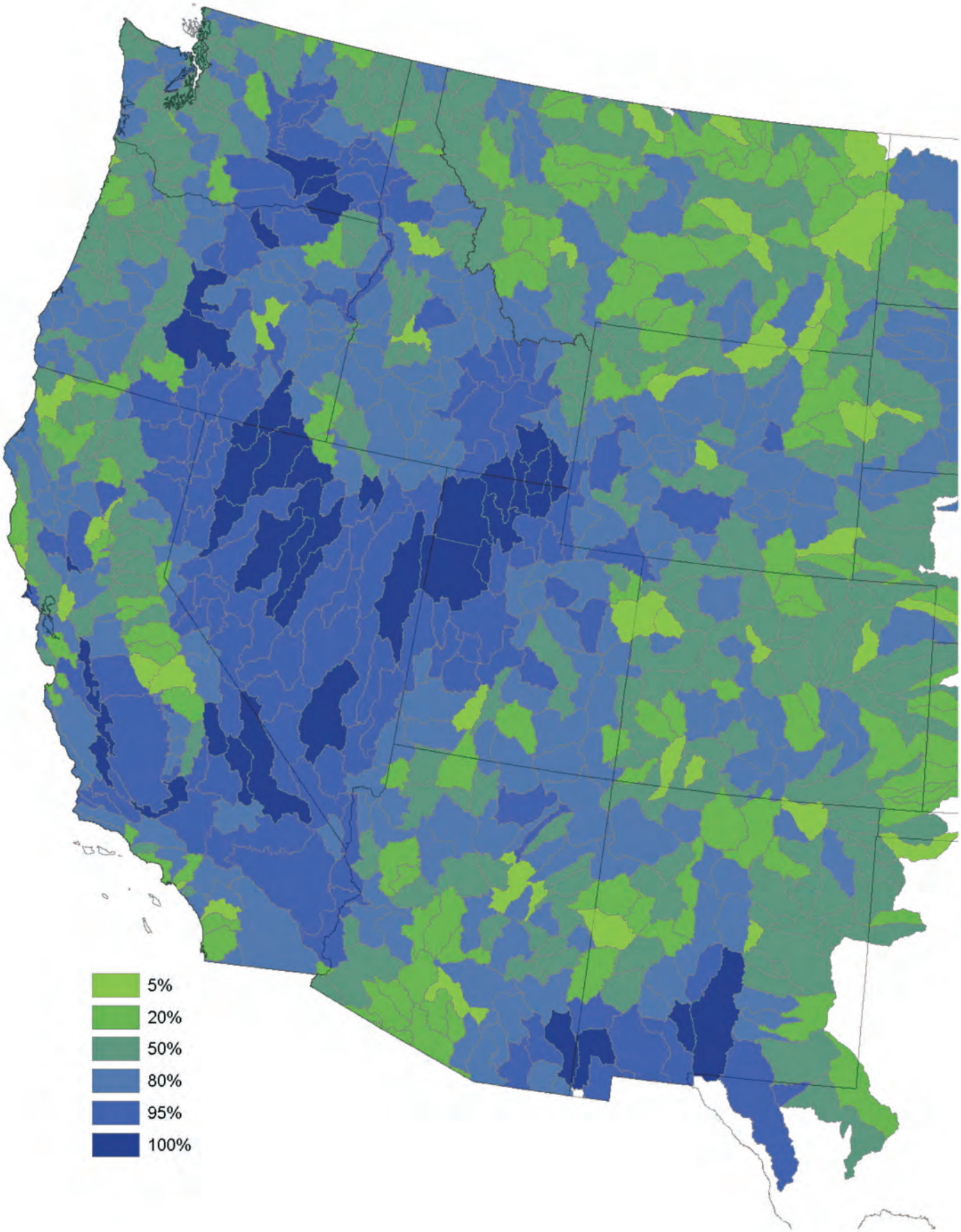

Figure 7(a). Absolute change in sedimentation from the past to current scenarios, averaged by watershed summarized by HUC8 watersheds. Those watersheds with the least change in sediment are assumed to be the best condition (top 5%; bright green), while watersheds with large changes from past ("natural") conditions are considered worse condition (bottom 5% or 95-100%; dark blue).

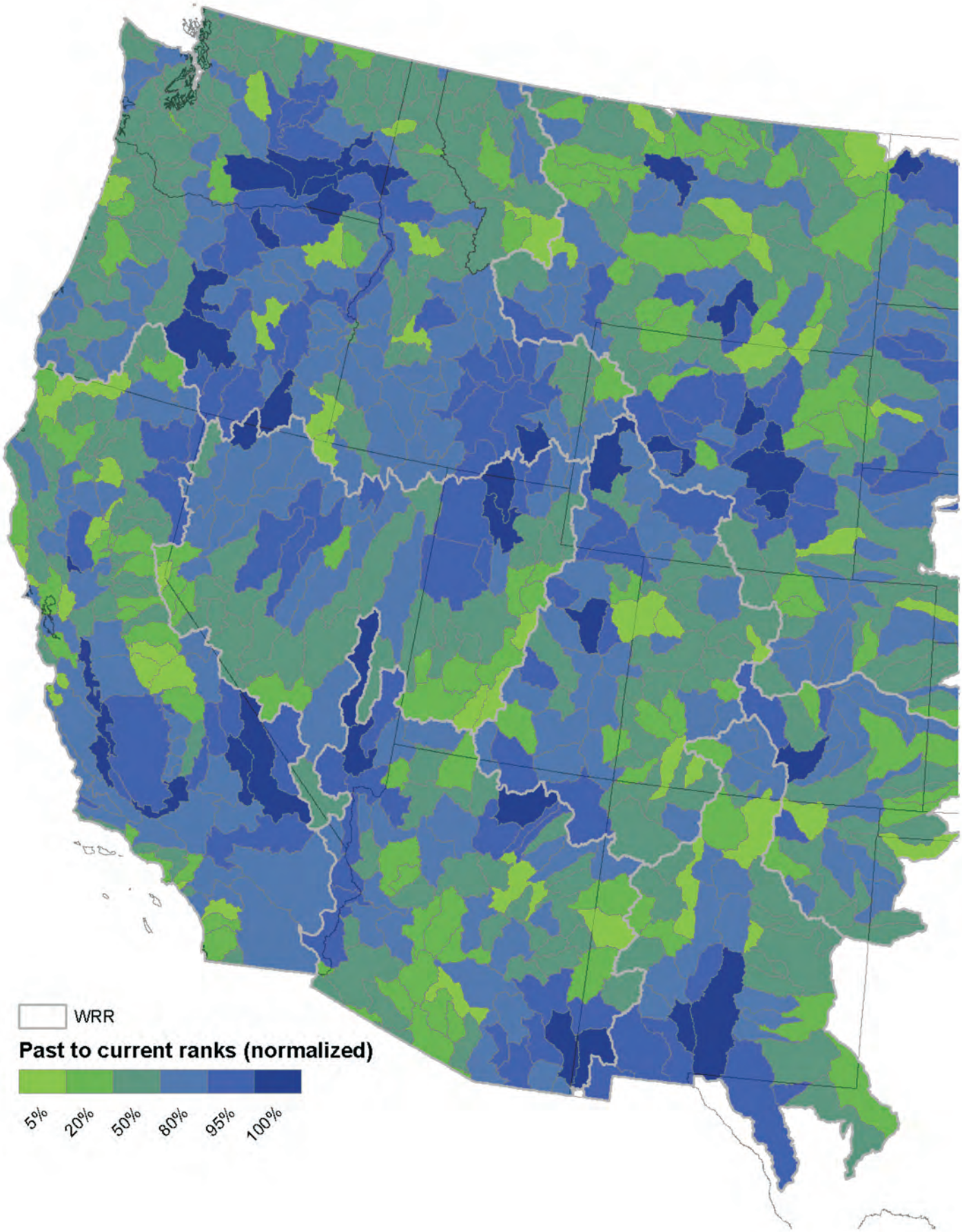

Figure 7(b). Absolute change in sedimentation from the past to current scenarios, averaged by watershed summarized by HUC8 watersheds and normalized. Those watersheds with the least change in sediment are assumed to be the best condition (top 5%; bright green), while watersheds with large changes from past ("natural") conditions are considered worse condition (bottom 5% or 95-100%; dark blue).

Major factors causing impacts

Sedimentation yield from a landscape is a function of the rate of chemical and physical weathering of parent material, the source of material in the form of soils, vegetation cover, the texture and characteristics of the soil, relief of the landscape, and the form and amount of precipitation for transporting the material. Through influencing vegetation cover and physical disturbance of soils, human activities have direct effects on sediment yield from landscapes. Livestock grazing, outdoor recreational vehicle use, agricultural activities, roads, urbanization, timber harvest, and other activities that influence vegetation cover and structure of the soils throughout the watershed can influence mobilization, transport, and delivery of sediment to stream channels. Some of these factors and their influences on stream channels can be mitigated to some degree through the presence of vegetation immediately adjacent to the channel and across the floodplain (Strayer et al. 2003, Langendoen, et al. 2009) and/or best management practices designed to reduce hillslope erosion (e.g., replanting, terracing, etc.). Sediment deprivation can lead to channel change as well.

Climate change implications

Through its influence on energy balance and the distribution of moisture, climate is perhaps the most important long-term factor influencing upland and riparian communities and ecosystems, as well as the fluxes of water and materials through watersheds (Bailey 1995, Benda and Dunne 1997). Decreases in precipitation and increases in temperature are anticipated to result in decreased vegetation cover and increased erosion and sedimentation throughout watersheds. Indeed, shifts from wet to a warmer and drier climate during the Holocene (2600 years before present), resulted in decreases in the extent of woodlands and increases in the dominance of desert scrub vegetation, resulting in significant increases in sediment yield from uplands to valley bottoms in the Great Basin (Chambers and Miller 2004). Dry climatic periods followed by wetter periods can result in significant erosion from landscapes. Under the climate change scenarios that we used in our sediment models, predicted sediment yield increased significantly throughout the West. Increases in sedimentation were particularly pronounced for the steeper and more arid portions of the western US and mountainous landscapes (e.g., southern Rocky Mountains, Basin and Range, Sierra Nevada). Shifts in form of precipitation from predominantly snow to rain due to warming winter temperatures also have implications for sediment yield from watersheds, leading to increases in surface flow and soil erosion. Sedimentation can have significant influences both directly on riparian areas through causing burial, aggradation, and erosion of fluvial surfaces across the floodplain and indirectly from sediment sources outside of the riparian zone. Sediment delivery to channels can occur through bank failure and channel related sources, from side slopes and uplands along valley slopes, and from tributaries and upstream sources. Climate-caused changes in rates of sediment yield from watersheds to river channels are confounded by (usually shorter-term) influences of human land-use activities on sediment erosion and transport rates.

Interacting factors

Separate and combined effects of climate, human activities, and land-use cover are difficult to ascertain at the scale of the western landscape, as some areas are likely to become wetter and cooler as others become drier and warmer. The combined effects of warmer and drier climate and intensification of human development (e.g., roads, ski areas, logging, and urban development) are likely to result in higher rates of sedimentation moving from watersheds into river channels. Drought, particularly drought following a wet period, may lead to severe fires, which inadvertently affect sediment yield from landscapes. These effects may be most severe immediately following fires, but may persist for centuries if intense fires result in permanent shifts in vegetation cover. Fires within riparian areas themselves have the confounding effect of removing vegetation that could potentially mitigate for increased sediment delivery to stream channels from uplands (Pettit and Naiman 2007).

The combined effects of warmer and drier climate and intensification of human development (e.g., roads, ski areas, logging, and urban development) are likely to result in higher rates of sedimentation moving from watersheds into river channels.

Drought, or prolonged hotter drier climate, may also cause changes in vegetation cover directly through increasing atmospheric demand for moisture and less available moisture in the soil and groundwater. Such changes can lead to decreased canopy cover of existing vegetation or shifts to completely different vegetation types (Merritt et al. 2010). Furthermore, as mentioned above, water stress in riparian areas can render individuals more susceptible to insect- and disease-caused dieback and mortality (Maxwell et al. 1997, Worrall 2009). Water stress was shown to be related to susceptibility of riparian *Alnus* to *Cytospora* canker in the southern Rocky Mountains (Worrall 2009). Similar patterns of dieback and increased vulnerability in upland vegetation throughout the watershed can pose a collective (e.g., additive or multiplicative) threat to riparian areas and stream channels (Negron et al. 2009).

One of our basic assumptions in this assessment is that any significant departure from the discharge and sediment volumes to which streams are currently adjusted, will result in ecologically significant shifts in the characteristics of riparian ecosystems. This assumption includes both significant decreases as well as increases from historic, reference conditions.

Increases in impervious cover on the landscape are likely to result in flashier hydrographs (higher peak and shorter duration) in stream channels. However, the decrease in total annual flow volume and seasonal peaks associated with snowmelt and monsoon hydrographs will affect the competence of stream channels to transport the increase in sediment volume entering channels (Schmidt and Wilcock 2008). Though difficult to generalize across streams, the combined effect of increased sediment yield from the landscape and decreased conveyance capacity in channels is likely to lead to reduced channel dimensions (widths and depths). Such factors are also likely to result in changes in floodplain and groundwater interactions with the channel and degradation of existing riparian ecosystems (viewed here as change form historic conditions). However, it is likely that these effects will vary across western ecoregions and for different valley and channel forms. Narrower, less extensive riparian areas resembling those along currently flashy, bedload dominated streams in the American southwest and Great Basin, may become more abundant in arid parts of the (future) West. One of our basic assumptions in this assessment is that any significant departure from the discharge and sediment volumes to which streams are currently adjusted, will result in ecologically significant shifts in the characteristics of riparian ecosystems. This assumption includes both significant decreases as well as increases from historic, reference conditions.

Changes caused to delivery of ecosystem services

Many of the ecological functions outlined above will also be affected by changes in sediment discharge to streams and rivers and subsequent changes in channel form and process. Buffering capacity, productivity, biological diversity, habitat values or riparian ecosystems will all shift from their historical state under altered sediment and flow regimes. Diminished buffering capacity coupled with increased sediment delivered to channels is likely to result in increased turbidity in streams as well as elevated concentrations of pollutants and suspended load, and a general decrease in water quality. Such degradation may diminish as a function of time, as it has during drier climatic periods throughout the Holocene (Miller et al. 2001). Soil formation is likely to slow under warmer, drier climates and sediment sources (principally upland soils) are likely to diminish as a function of time as well. The characteristics of streams are likely to shift northward and to higher elevations under each of the climate change scenarios presented here. Constraints on such shifts include the efficiency of corridors and the mobility of organisms (presumably north south oriented rivers should experience more rapid adjustments) as well as geological constraints (e.g., narrower valleys with less extensive floodplains as conditions migrate to higher elevations).

Management implications

There is a range of management activities that influence sediment yield from watersheds to rivers. Best management practices, can minimize the generation and transport of sediment during development activities. Sediment fences, culverts, terracing, and manipulating vegetation cover are all very effective means

of regulating sediment yield from hillslopes. Fire management through fuels treatments, thinning, controlled burns and other means are ways in which fire severity and its influence on soils and standing vegetation can be affected by management activities. The density, orientation, proximity, and configuration of roads, the construction of bridges and culverts, and drainage management all exert important influences on sediment yield (Trombulak, and Frissell 2000).

Controlling other factors in riparian areas themselves may also help to mitigate for increased sediment yield from uplands. Among these are maintenance of sufficient cover and extent of riparian vegetation through preservation and/or through active or passive riparian restoration (Wohl et al. 2005). Reducing a number of other stressors, such as livestock or wildlife grazing or browsing intensity, recreation and outdoor recreational vehicle (ORV) use, development on floodplains and in riparian areas, and other factors that directly influence riparian vegetation or the processes that support it, can enable riparian areas to sustain more frequent, higher intensity, or more sustained external stress levels without resulting in as severe degradation or response.

3. Riparian zone/valley confinement

Overall, we found that 14.8% of the West's potential riparian areas (available habitat in valley bottoms) are modified by roads, development, or agriculture (cropland/pastureland, not grazing). This will likely increase to 15.8% by 2030 due to land use encroachment associated with housing development (but does not include changes to the transportation infrastructure). By water region, development of potential riparian zones is largest in California, and will likely increase by up to 50 to 100% for most regions (Figure 8; Table 6). We also estimated the proportion of different types of natural cover types in the riparian zone (Tables 7 & 8), as well as the percent occupied by roads and agriculture (Table 9).

The watersheds with riparian zones that have most been most heavily modified include the Central Valley and Los Angeles basin of California, the Willamette Valley, Oregon and eastern Washington, and northern Montana (Figure 9). Watersheds in the southern Sierra, the northern Rockies in Idaho, the Colorado plateau, and eastern Wyoming are in the best condition.

Table 6. The percent of potential riparian areas modified by human-dominated land uses including urban, roads, and cropland agriculture.

WRR	Current	Future (2030)
Arkansas (11)	11.7	12.0
California (18)	17.9	20.5
Colorado (14-15)	6.3	7.3
Great basin (16)	14.0	15.5
Missouri (10)	19.7	19.9
Pacific NW (17)	16.7	17.4
Rio Grande (13)	7.0	7.4

Reducing a number of other stressors, such as livestock or wildlife grazing and browsing intensity, recreation and outdoor recreational vehicle use, development on floodplains and in riparian areas, and other factors that directly influence riparian vegetation or the processes that support it, can enable riparian areas to sustain more frequent, higher intensity, or more sustained external stress levels without resulting in as severe degradation in response.

T*able 7. Past area (km^2) of "natural" cover types within the modeled potential riparian zones (using LANDFIRE Biophysical Settings).*

WRR	Forest	Shrubland	Grassland
Arkansas (11)	2,780	2,006	13,398
California (18)	30,956	21,888	3,783
Colorado (14-15)	15,517	64,963	756
Great Basin (16)	7,163	48,392	756
Missouri (10)	39,002	21,996	62,110
Pacific (17)	34,750	34,750	948
Rio Grande (13)	5,353	13,208	12,985

The watersheds with riparian zones vegetation that have been most heavily modified include the Central Valley and Los Angeles basin of California, the Willamette Valley, Oregon and eastern Washington, and northern Montana.

Table 8. Current area (km^2) of "natural" cover types within the potential riparian zones (using LANDFIRE Existing Vegetation Types).

WRR	Forest	Shrubland	Grassland
Arkansas (11)	1,292	2,208	9,518
California (18)	11,191	15,036	938
Colorado (14-15)	13,646	60,941	4,810
Great Basin (16)	4,165	42,048	1,291
Missouri (10)	16,832	16,680	50,016
Pacific NW (17)	18,542	22,889	1,585
Rio Grande (13)	3,072	20,650	4,900

Table 9. The percent of potential riparian areas currently occupied by roads and agriculture.

WRR	All Roads	Secondary	Highways	Agriculture
Arkansas (11)	7.9	0.6	1.3	10.3
California (18)	1.8	1.4	0.4	10.1
Colorado (14-15)	1.1	1.0	0.1	6.2
Great Basin (16)	1.3	1.0	0.3	13.6
Missouri (10)	1.0	0.8	0.2	16.0
Pacific NW (17)	1.1	0.9	0.2	13.2
Rio Grande (13)	0.8	0.7	0.1	7.1

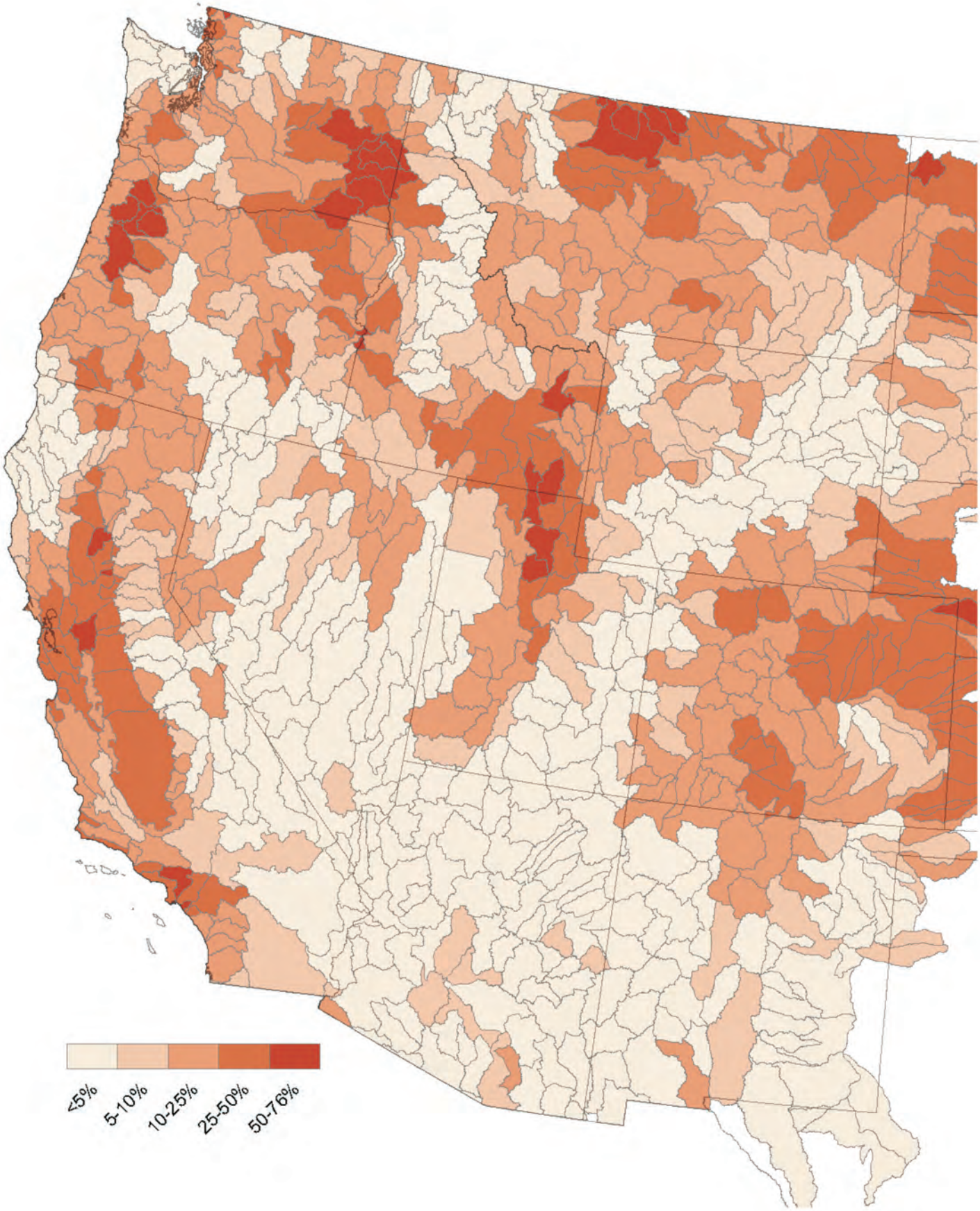

Figure 8(a). The percentage of potential riparian zone modified by development, roads, or agricultural (cropland/pastureland), averaged for each watershed (HUC8) for the current scenario.

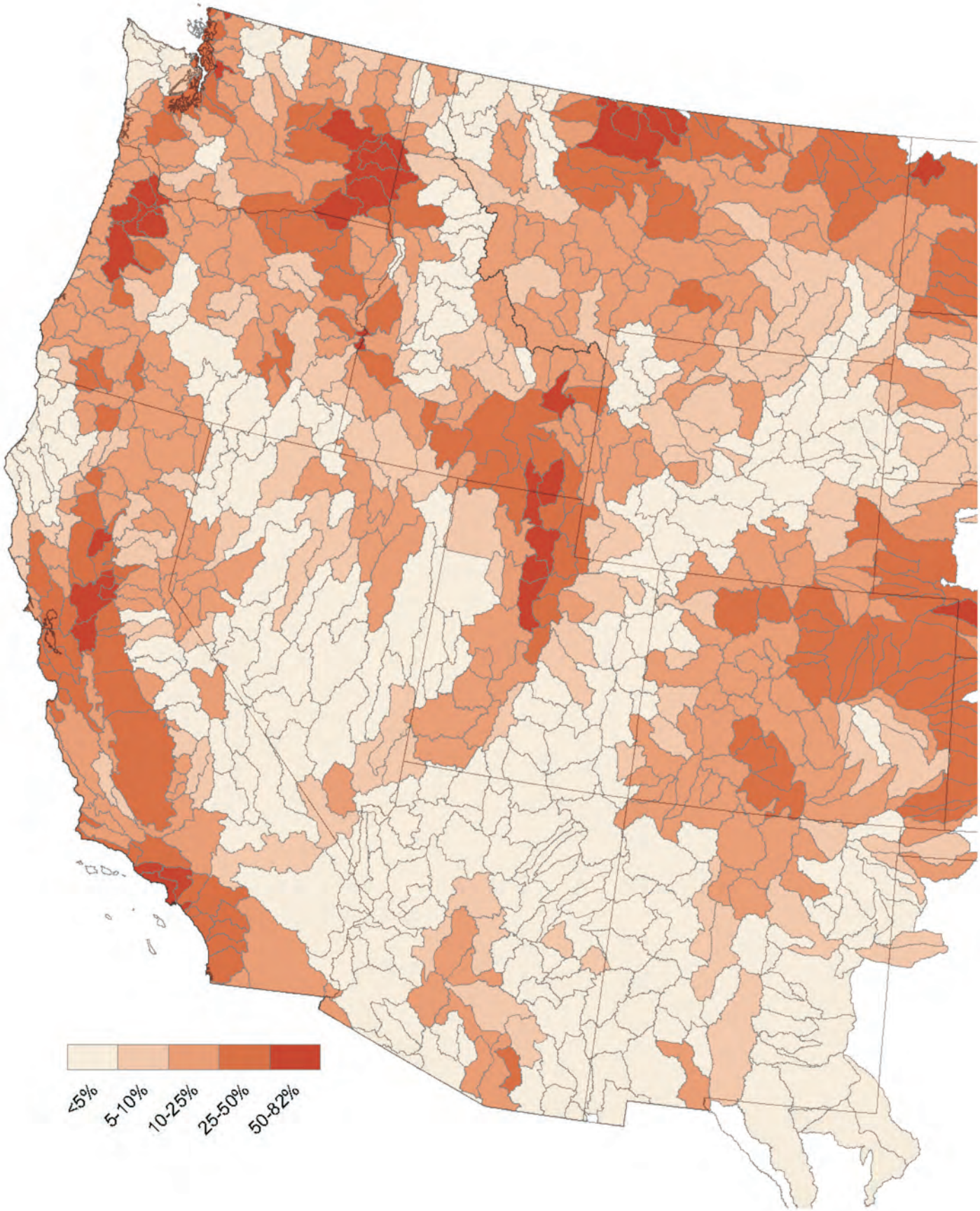

Figure 8(b). The percentage of potential riparian zone modified by development, roads, or agricultural (cropland/pastureland), averaged for each watershed (HUC8) for the forecasted future.

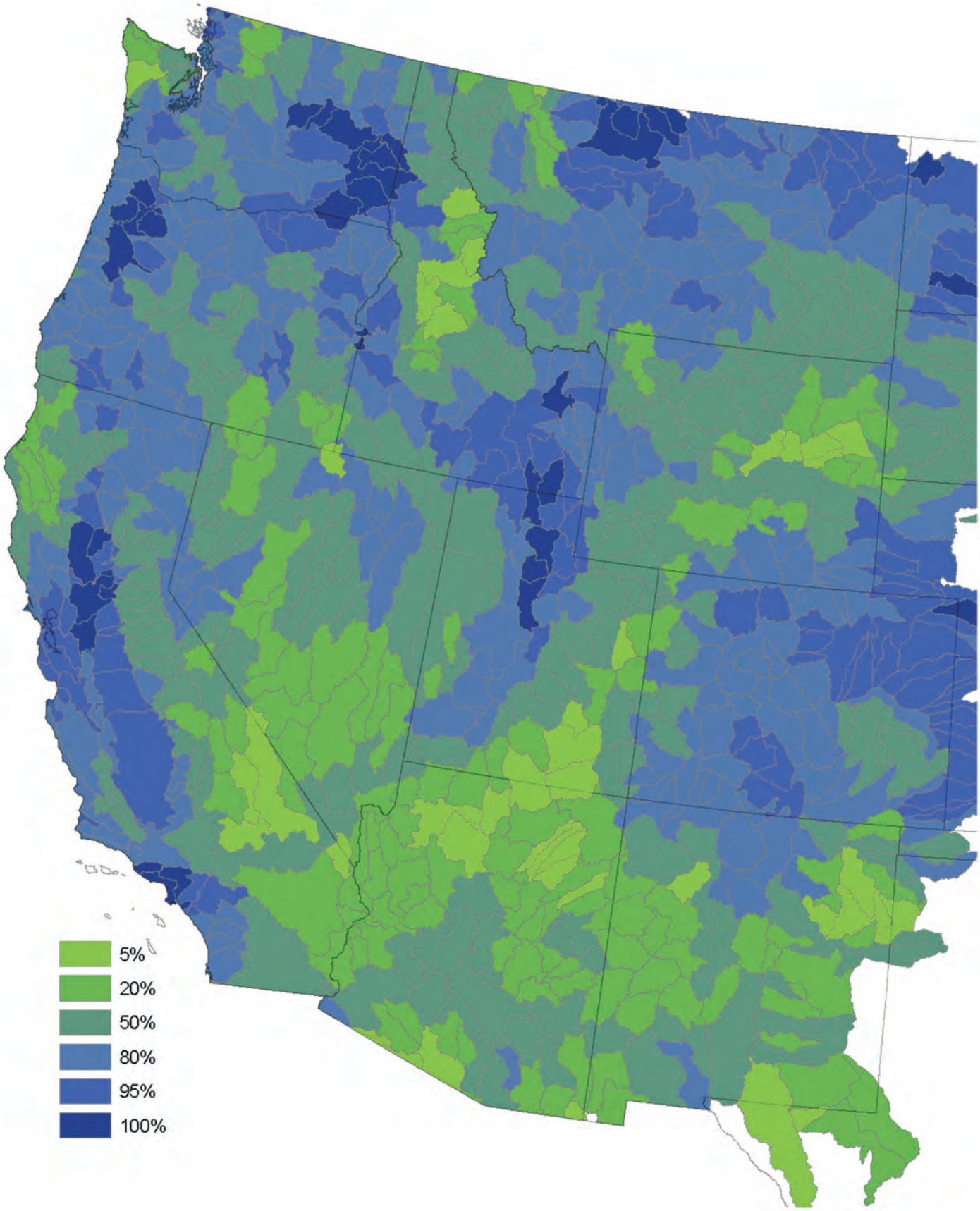

Figure 9(a). Watersheds that contain the least (green) and most (blue) modified potential riparian zone for the current scenario. Note that the Colorado (WRR 14 & 15) is mapped as one region.

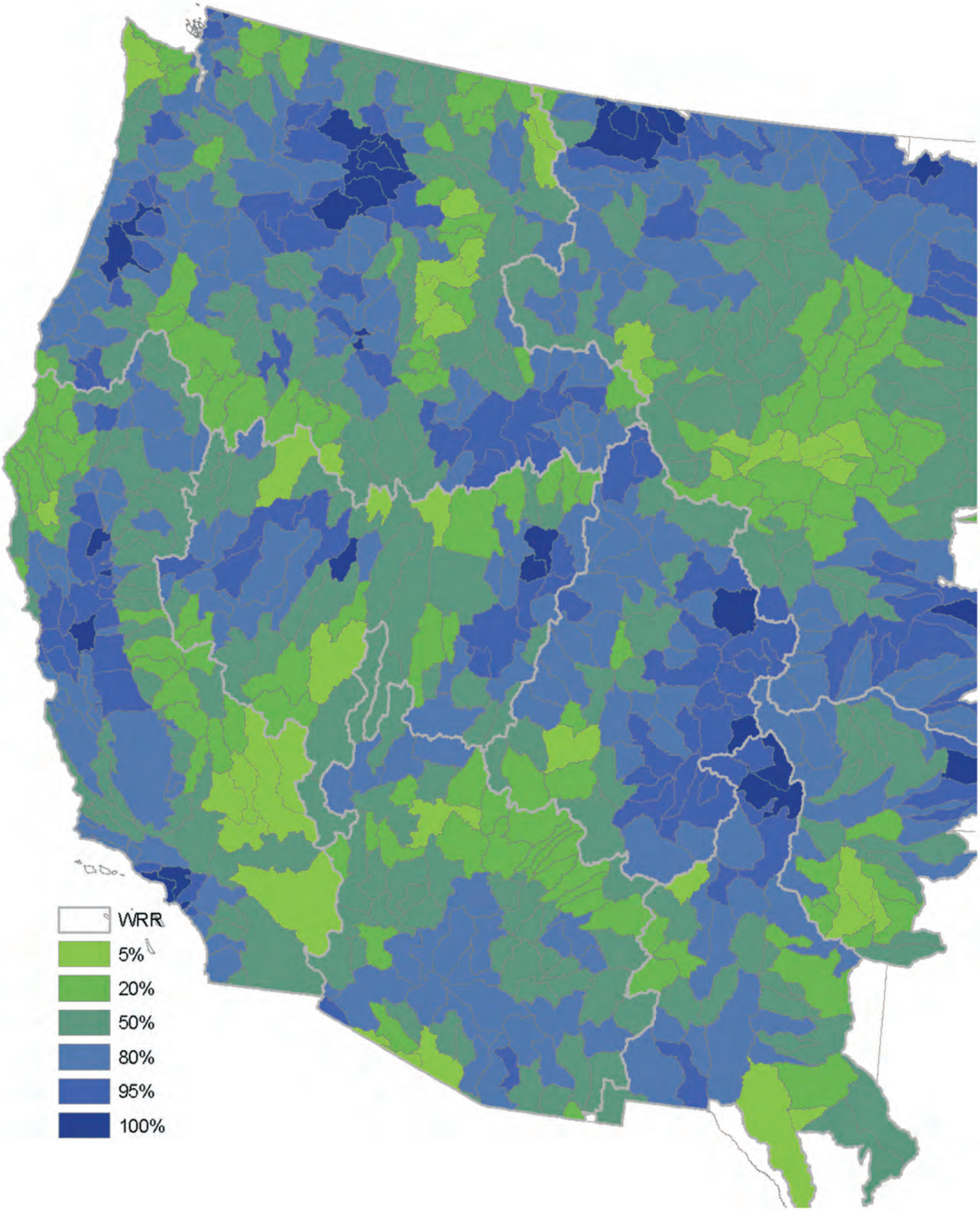

Figure 9(b). Watersheds that contain the least (green) and most (blue) modified potential riparian zone for the current scenario, normaliezed by region. Note that the Colorado (WRR 14 & 15) is mapped as one region.

Major factors causing impacts

Rivers have been deliberately confined to narrower channels through levee construction, channelization, and altered river flow regimes for centuries Ninety percent of floodplain riparian forests in the eastern US are functionally extinct due to channelization, flow-related loss of lateral connectivity, and encroachment by human development in floodplains (Tockner and Sanford 2002). Continued permitting of building in floodplains by cities and counties will continue to result in less cover of riparian area than 'potential riparian area' based upon valley width/confinement. Whereas roads, bridges, pipelines and other infrastructure may allow for flooding of floodplains, these structures also limit the area of valley bottom that can function as historically and support riparian vegetation typical of historic conditions such as marsh, wet meadow, and bottomland forest. The use of valley bottoms for agricultural activities, golf courses, and parks and open space allows for flooding as well as natural functioning riparian areas in some circumstances, however, damage to greens, crops, and irrigation infrastructure still creates conflict.

Climate change implications

In our analysis, valley confinement represents the potential extent of riparian areas in valley bottoms. The actual (or realized) extent of riparian areas expands and contracts within this potential. Under altered flow and sediment regimes projected by future climate change scenarios and sediment modeling, realized riparian area is dramatically smaller than potential for most watersheds in the western US. Though lithologic constraints on valley bottoms change imperceptibly over geologic time, the influences of climate change on water delivery to valleys, stream discharge, groundwater volume and seasonal fluctuations, and atmospheric demand for moisture, result in changes in riparian width over decades. Interactions between discharge and valley bottoms are important as they influence hydraulics and are conditioned by climate change. For example, the same flood may have higher shear stress, stream power and transport capacity in a narrow confined valley, with consequences to channel form and riparian vegetation (Nanson and Croke 1992). Higher discharges caused by increased snowmelt associated with climate change may have a greater influence on confined valleys compared to wider less, confined valleys. Furthermore, the influence of sideslope processes from valley walls on the channel itself could be influenced by climate change through altered rates of hillslope failure, debris flow, tributary sediment inputs, and stream competence to transport inputs of material to the channel (Benda and Dunne 1997).

Interacting factors

The influence of sideslopes and valley walls varies as a function of degree of valley confinement and process domain (*sensu* Montgomery 1999). Typically, headwater streams are more directly influenced by side slope processes (e.g., colluvial processes such as landslides and rockfall) and sideslope processes diminish in importance as a function of downstream distance or stream order. Of course, an exception is canyon segments of larger order streams (e.g., Snake, Salmon, and Colorado Rivers). The influence of these processes on riparian form and functioning is important but varies as a function of the realized riparian extent and proximity of the channel to valley sidewalls (Friedman et al. 2006).

Changes caused to delivery of ecosystem services

Through their influence on the potential width and aerial extent of riparian areas, valley confinement and sideslope processes exert tremendous influence over the quantity and characteristics of ecosystem services associated with riparian areas. Greater aerial extent of riparian areas serves to buffer streams and aquatic ecosystems from watershed and floodplain processes. In addition, greater extent of riparian areas provides more area for recreational opportunities, groundwater storage, vegetation and associated riparian habitat, provisions of allochthonous inputs and woody debris, to freshwater ecosystems, flood attenuation capacity, nutrient

processing, and myriad other benefits. The area over which riparian ecosystems exist is constrained by valley confinement, thus any ecosystem functions performed by riparian areas are constrained as well. Encroachment by human activities is fundamental threat to riparian structure and function. Human activities and development on floodplains and across valley bottoms will increase as a function of human populations in the western US, influencing both the types and qualities of ecosystems provided by river bottomlands. Tockner and Stanford (2002) rank North American river floodplains among the most threatened globally, second only to southeast Asia and Sahelian Africa.

Human activities and development on floodplains and across valley bottoms will increase as a function of human populations in the western US, influencing both the types and quality of ecosystems provided by river bottomlands. Tockner and Stanford (2002) rank North American river floodplains among the most threatened globally, second only to southeast Asia and Sahelian Africa.

Management implications

Because riparian areas naturally occupy a small proportion of the total land cover even in the wettest parts of the western US (e.g., Pacific Northwest, parts of the Rocky Mountains and Sierra Nevada) and perform disproportionately high levels of ecosystem services, prevention of encroachment and subsequent loss of riparian areas is a fundamentally important management goal. Prevention of development in floodplains, establishment of riparian buffers, management of human activities, livestock grazing, weed and vegetation management in general is advisable. Management of factors that have the potential to compromise the function of riparian areas is important as well. Management of headwaters and streams and uplands tributary to a reach of particular management interest is key to effective local management. Education of private land owners about the importance of riparian buffers, of appropriate livestock management techniques, and management of cultivated lands and proper application of insecticides, herbicides, and fertilizers can have tremendous influence on vegetation health and nutrient and chemical discharge to rivers at scales from the reach and segment to the watershed.

Synthesis of factors into the riparian threats score

We standardized each individual factor considered (hydrologic, sediment, and valley confinement) and then summed their values to calculate a single, integrative index called the “riparian threats score”. This threat score provides a relative index of threats to riparian systems.

Threats in the past to future scenarios (Figures 10-12) indicate that the highest threats westwide occur largely in western Washington, Great Basin, southern Idaho and northern Utah, and southern Arizona and New Mexico. The least threatened include parts of the Cascade and Sierra ranges and eastern and southern Utah and western Colorado. To provide more detailed information at a finer resolution, we mapped the riparian threats score at the reach catchment (HUC12) level as well (Figures 12).

Threats in the past-to-future scenarios indicate that the highest threats westwide occur largely in western Washington, Great Basin, southern Idaho and northern Utah, and southern Arizona and New Mexico. The least threatened include parts of the Cascade and Sierra ranges and eastern and southern Utah and western Colorado.

Because there are no comprehensive datasets readily available that provide field-based riparian condition estimates it is difficult in general to provide a rigorous validation of our riparian threats score. However, we did compare our results to the Environmental Protection Agency’s (EPA) Wadeable Stream Assessment (USEPA 2006) and found that the riparian threats score is generally consistent with the EPA’s findings. That is, the average riparian threats scores for the reference sites were better than the “stressed” sites, at both HUC8 and HUC12 scales. The average score (by HUC8) for reference sites was 0.626 (SD=0.835) as compared to 0.511 (SD=0.580) and 0.681 (SD=0.709) for the “moderately stressed” and “stressed” sites. The average score by reach catchment area (HUC12) for reference sites was 1.087 (SD=2.426) as compared to 0.791 (SD=2.039) and 2.000 (SD=2.984) for the “moderately stressed” and “stressed” sites. Note that the “moderately stressed” sites do not conform to our expectations, as their condition is consistently better than the reference sites. In total, there were 1,582 WSA sites in our study area.

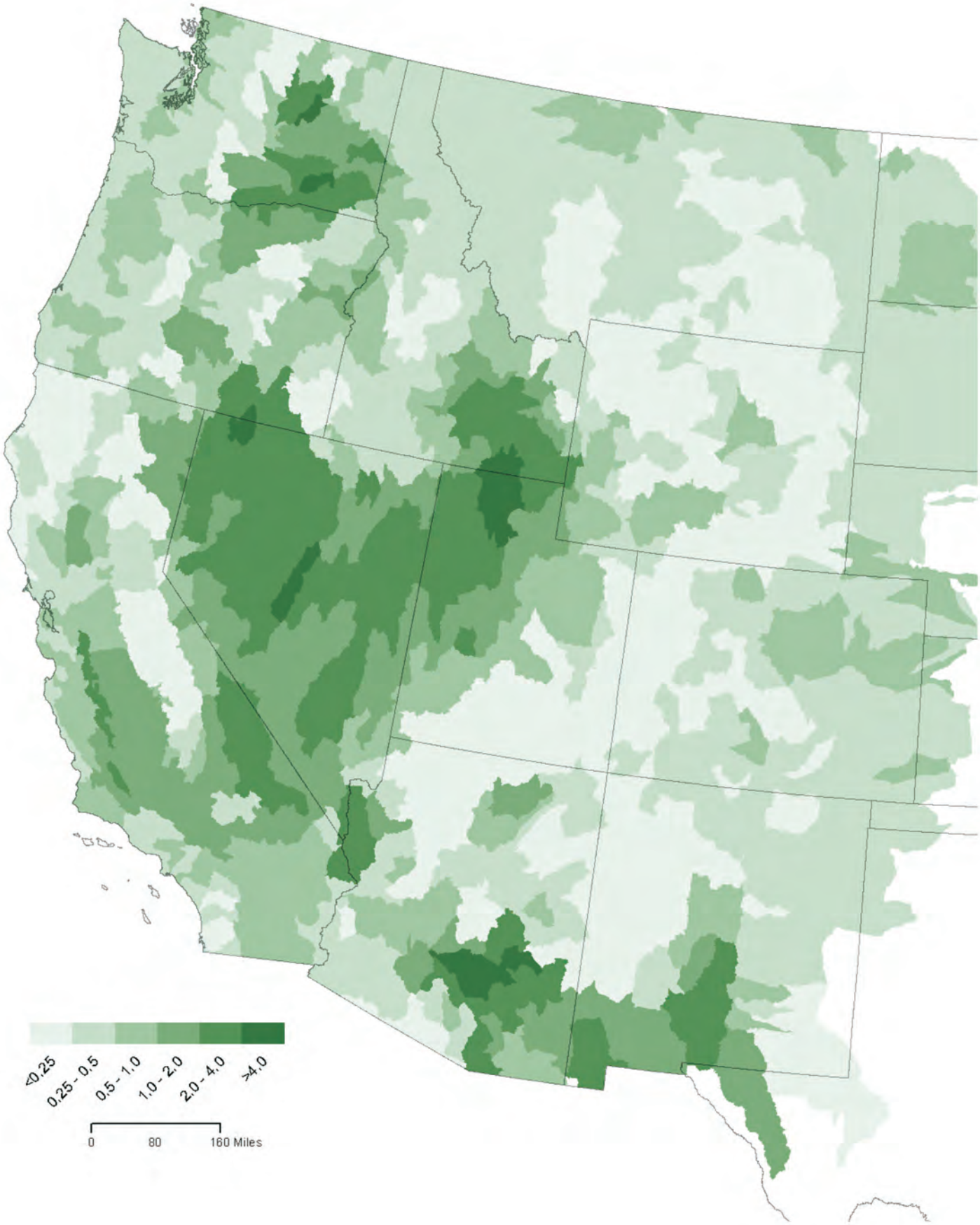

Figure 10(a). The raw values of riparian threats score for the past to current scenarios. Score values were calculated by standardizing the flow fragmentation, sediment, and valley confinement values and then summing to score areas with the greatest threat level (dark green) through lowest threat level (light green).

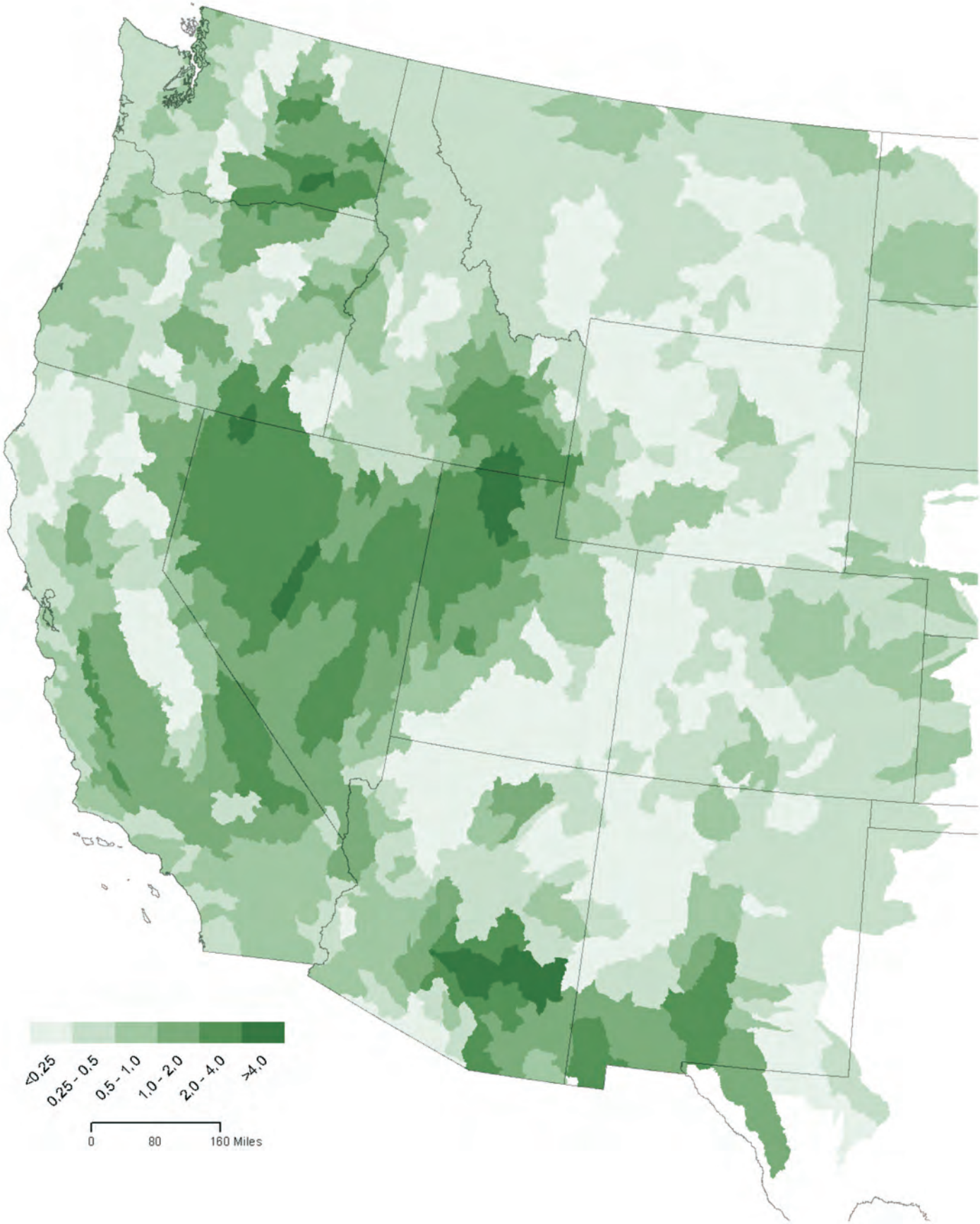

Figure 10(b). The raw values of riparian threats score for current to A1B future scenarios. Score values were calculated by standardizing the flow fragmentation, sediment, and valley confinement values and then summing to score areas with the greatest threat level (dark green) through lowest threat level (light green).

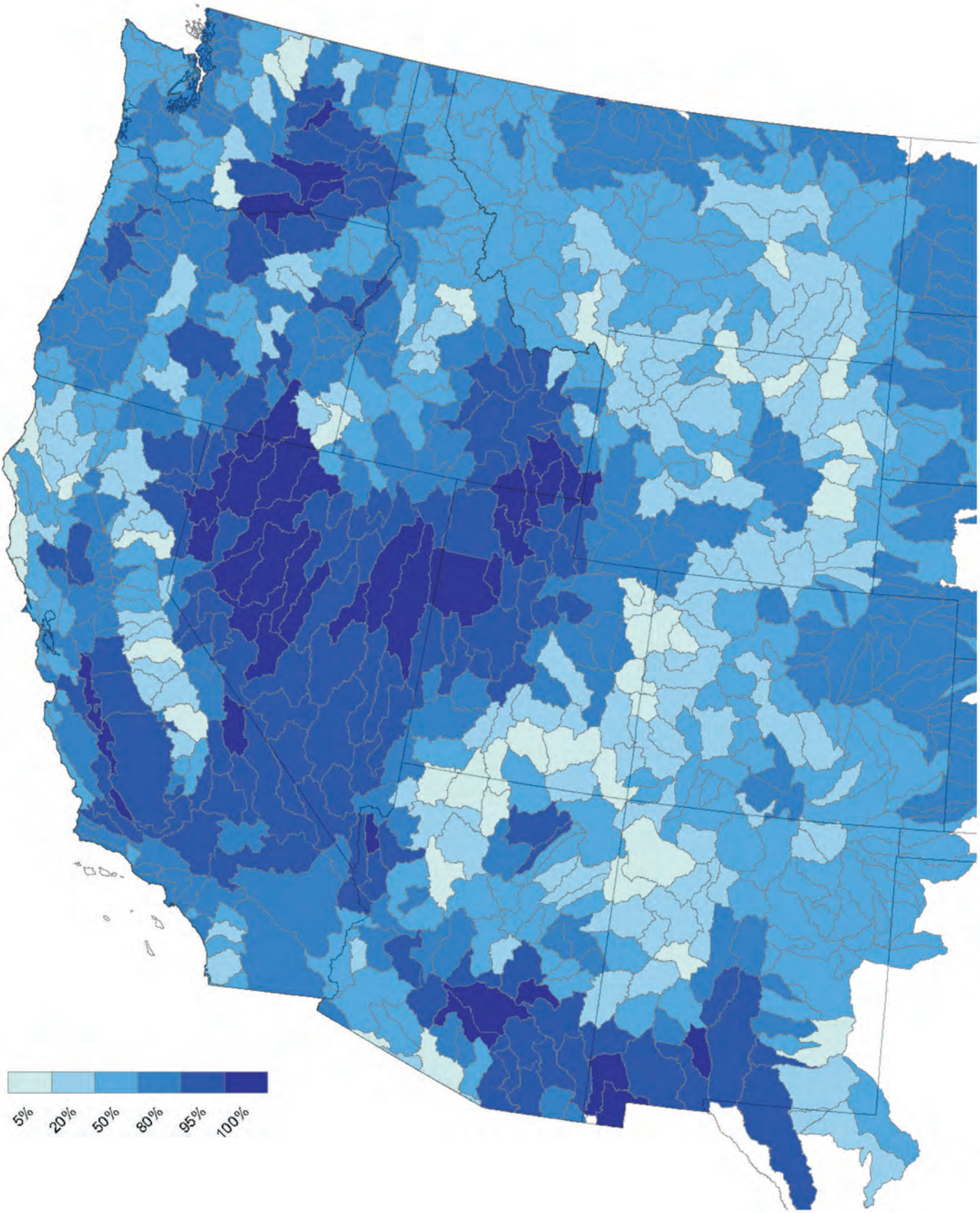

Figure 11(a). The riparian threats score for the past to current scenarios. The score was normalized to show the highest threat level (dark blue) through lowest threat level (light blue) using percentiles.

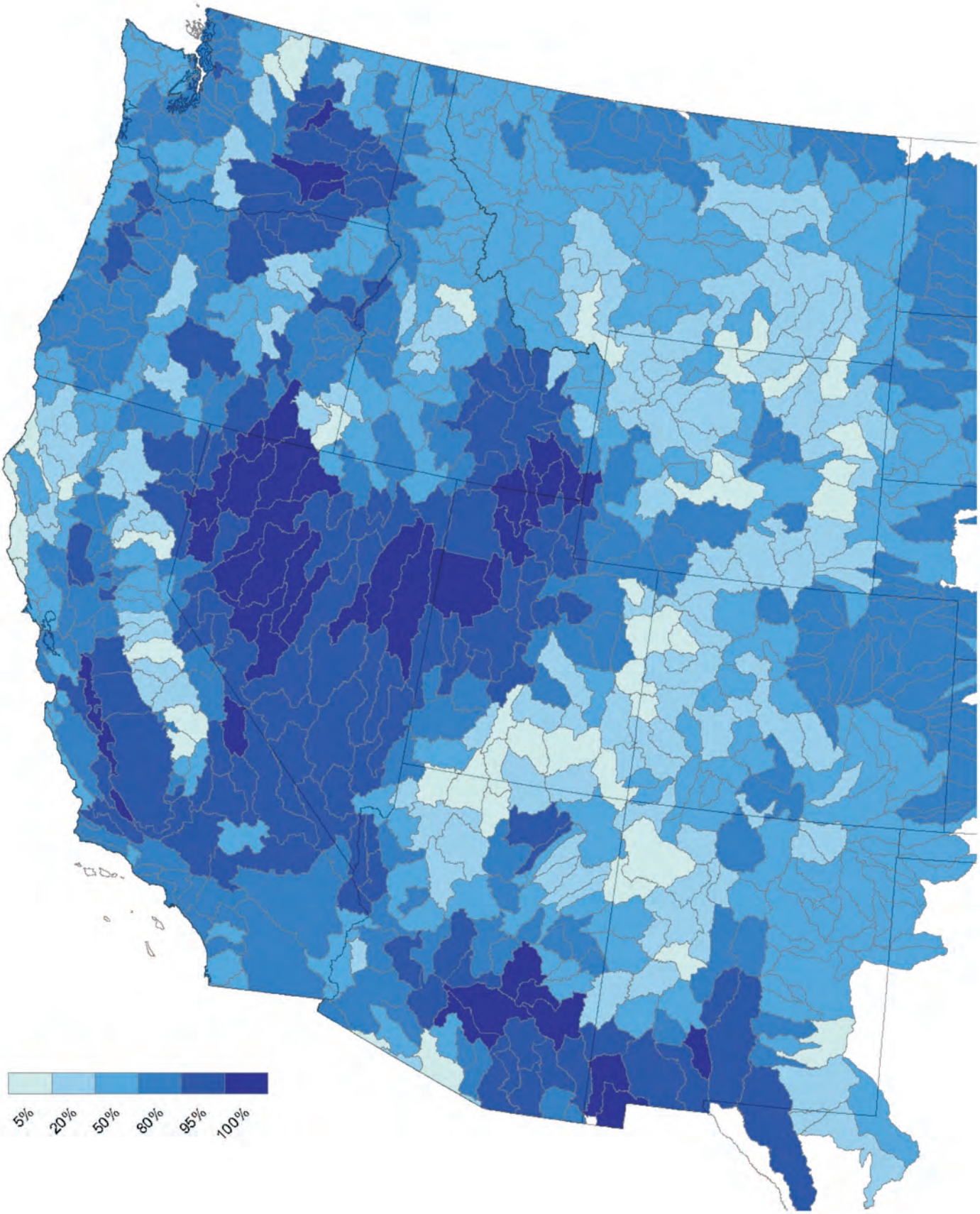

Figure 11(b). The riparian threats score for current A1B scenarios. The score was normalized to show the highest threat level (dark blue) through lowest threat level (light blue) using percentiles.

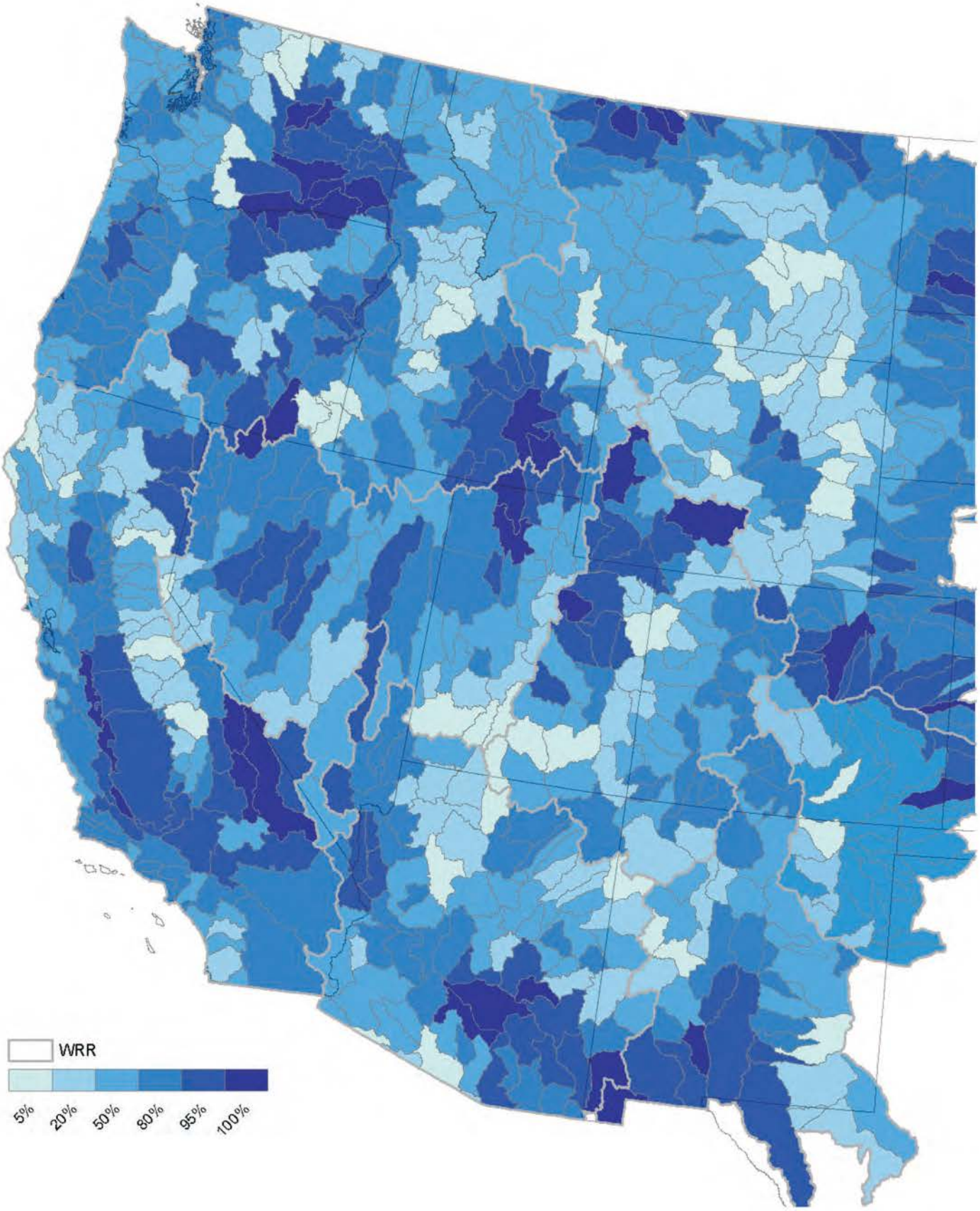

Figure 12(a). Riparian threats score, past to current scenarios, normalized by water resource region.

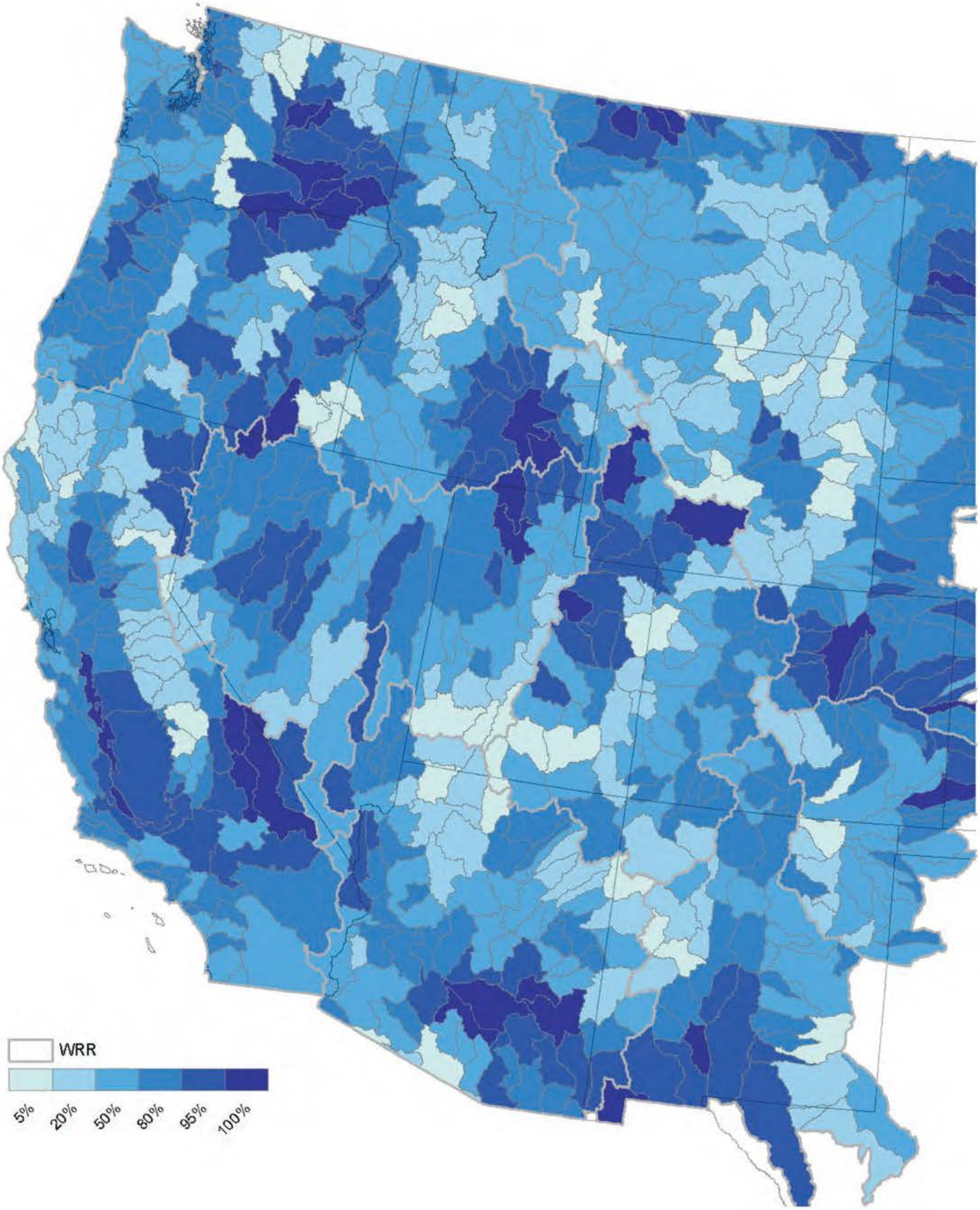

Figure 12(b). Riparian threats score, current to A1B future, normalized by water resource region.

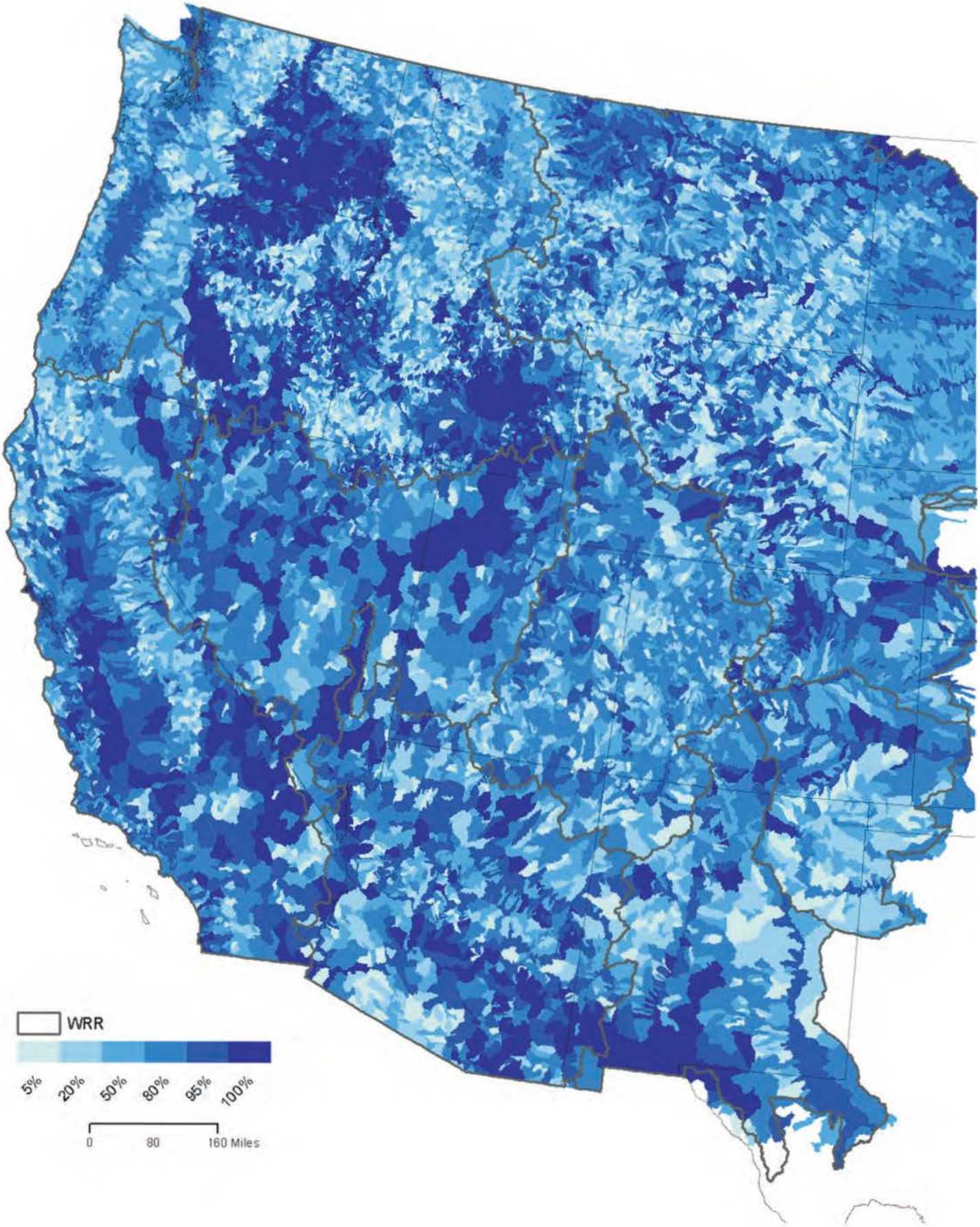

Figure 13. Riparian threats score for reach catchment areas (~HUC12), normalized by water resource region.

Summary

Climate change, increased human demands for water, continued water development and their combined and interactive effects pose significant threats to native riparian habitats throughout the West (Baron et al. 2002, Wohl 2005). Expansion and continued operation of hydropower and micro-hydropower facilities will continue to regulate the flow of rivers as human demands for clean sources of energy intensify (EPA 2005). For example, the Bureau of Reclamation (BuRec) currently generates 40 billion kilowatt hours of electricity annually from its 58 hydropower facilities. There are approx. 200 FERC hydropower projects on US Forest Service Lands, generating approx. 16,500 megawatts annually; an additional 1,000 megawatts are generated by 71 private power plants. BuRec is currently conducting cost benefit analyses on 530 potential hydropower sites in 17 western states (EPA 2005). The ecological consequences of reduced stream flow associated with water storage, extraction, or altered flow regimes for storage, flood control or hydropower generation include reduced channel capacity and aquatic habitat, encroachment of upland vegetation into riparian areas, reduced extent of riparian zones, and drought stress-related shifts in the distributions of plant and animal populations and communities.

The ecological consequences of reduced stream flow associated with water storage, extraction, or altered flow regimes for storage, flood control or hydropower generation include reduced channel capacity and aquatic habitat, encroachment of upland vegetation into riparian areas, reduced extent of riparian zones, and drought stress-related shifts in the distributions of plant and animal populations and communities.

Overall (see Table 10), we found that the average flow modification indicates that reservoirs can store between 16% to 200+% of the annual stream discharge delivered to streams, but is highly variable with lower modification for the regions of the Pacific Northwest (~18%) and the upper Missouri (~47%), and higher modification in the Rio Grande (~272%), Great Basin (~225%), and Colorado (~220%) basins. We also found that 14.8% of the West's potential riparian areas are modified by roads, development, or agriculture (cropland/pastureland, excluding grazing).

Table 10. Summary of the main findings of the three process-factors for riparian threats.

	LONGITUDINAL Flow fragmentation	UPLAND Sedimentation	RIPARIAN ZONE Proportion modified
Currently	Ranges from 16 to 274% of mean annual flow	High increases in sediment production in the Great Basin, eastern Washington, and southern New Mexico Marginal declines in sediment production in urbanizing areas	Ranges from about 7% (Colorado and Rio Grande WRRs) to 17%, (California, Pacific NW, and Missouri)
Current to future	Declines in the Arkansas & Rio Grande WRR; strong increases in California & Colorado WRR	Similar patterns with small increases in the decline of sediment production due to forecasted urbanization	Will likely increase due to forecasted growth, especially in California and Colorado WRRs by up to 3%

Table 11. The average riparian threats score averaged by major land owner/manager in the West, sorted from highest average value to lowest.

Owner	Area (km^2)	%	Past to Current (Mean & SD)		Current to future (Mean & SD)		% Increase
Department of Defense	67,085	2.0%	1.47	0.92	1.54	0.92	4.5%
Bureau of Reclamation	4,057	0.1%	1.41	1.36	1.44	1.56	2.3%
Fish and Wildlife Service	29,012	0.8%	1.37	1.65	1.42	1.68	3.8%
Bureau of Land Management	682,848	20.0%	1.21	1.12	1.25	1.17	2.8%
Non-government org. - conserv.	5,690	0.2%	1.02	1.08	1.07	1.13	5.5%
Public local	4,953	0.1%	0.87	0.78	1.04	1.08	19.9%
National Park Service	78,380	2.3%	0.80	0.94	0.82	0.95	1.4%
State	172,387	5.0%	0.78	0.95	0.86	1.17	11.1%
Private	1,584,410	46.4%	0.67	0.78	0.69	0.81	3.4%
Native American	184,884	5.4%	0.59	0.84	0.69	1.20	18.2%
Forest Service	584,800	17.1%	0.52	0.63	0.57	0.73	8.7%
Other Federal	18,443	0.5%	0.50	0.40	0.53	0.43	5.5%
Unknown	2	0.0%	0.35	0.05	0.39	0.01	11.5%

Limitations and gaps

It is important to stress that because detailed, field-based measurements regarding each of the factors included in our analyses are unavailable at the scale of river reaches or segments throughout the West, there are a number of important caveats and limitations to our findings. Our assumption is that quantifying the factors which govern these processes (and for which there is reliable information) provides the best basis for assessing riparian conditions relative to historic conditions, to detect patterns and trends across the western US, and to project likely future changes tied to human and natural changes conditioned by land use and climate.

Findings from our analyses should serve as a red flag for areas in the western US that have the potential to change most dramatically in the future due to human and climate-caused change (i.e. those with high riparian threats scores).

In general, we found little difference between the threat indicators calculated based on the two different forecasted climate change scenarios. However, because only the flow fragmentation measure was directly linked to changes in climate, and because of the relatively coarse aggregation at HUC8 level, our effort was relatively insensitive to potential differences that might be observed with more detailed geospatial information.

We identified early in the project the need to conduct analyses on the implications for dam storage increases based on population growth, changes in fire regimes and pest outbreak (e.g., pine beetle) that might affect sediment production and delivery to riparian areas, and invasive species such as tamarisk. However, we did not conduct these analyses because appropriate data on these threats were not available.

Findings from our analyses should serve as a red flag for areas in the western US that have the potential to change most dramatically in the future due to human and climate-caused change (i.e. those with high riparian threats scores).

Areas with a high threat rating in our analyses are those that warrant further and more detailed analyses at a finer spatial scale. As with any large scale analyses, the rankings and maps presented here should be interpreted with caution and with full understanding of the factors utilized in our work. In the future, more detailed analyses will become possible as more extensive, continuous, consistent, and higher resolution geospatial data become available.

Conclusions

It is important to recognize that along with continued human demands for water, timber, recreational opportunities, development and agricultural opportunities along rivers, floodplains, and across valley bottoms come many opportunities to improve management of riparian areas. Through recognition of the hierarchy of threats to riparian ecosystems, from regional to local, we are better equipped to manage, sustain and enhance the ecological services provided by rivers

Fundamentally, the physical integrity of river channels and floodplains provides the template for healthy riparian ecosystems through maintaining the form and processes that support them (Graf 2001). River hydrology and associated hydraulics and fluvial processes that support riparian vegetation and drive many of the ecological services provided by rivers are therefore are the focus of this riparian threats assessment.

Fundamentally, the physical integrity of river channels and floodplains provides the template for healthy riparian areas through maintaining the form and processes that support them. River hydrology and associated hydraulics and fluvial processes, longitudinal connectivity along rivers, and lateral exchanges of water, carbon, and materials are the key processes and linkages that support many of the ecological services provided by rivers - and therefore are the focus of this riparian threats assessment.

Recognition that vegetation directly or indirectly provides a vast majority of ecosystem services associated with riparian areas is an important step in managing riverine and riparian ecosystems. Hydraulic roughness and bank stability, nutrient uptake, inputs of carbon and nutrients (and large wood), habitat (forage, cover, nesting), linkages between vegetation, hydrology, sedimentation and channel form. Managing the factors that influence vegetation and support desired ecosystem services requires an understanding of the reciprocal linkages between vegetation, hydrology, sedimentation and channel form. Because the fundamental processes influencing riparian vegetation are related to flow (hydrology and hydraulics) and channel processes and form (governed by valley form, flow and sediment delivery and transport), we chose to focus this threats assessment on a characterization of the historic, current and future rates and volumes of these factors. There is an extensive literature and over a century of adaptive management in the western US that ties land use practices and long term climatic patterns to each of these factors both within riparian areas and throughout the watersheds that influence them. There are also some recent tools for river management that, as they are developed and tested, will be very effective in managing rivers and their riparian ecosystems (Poff et al. 2010).

Managing the factors that influence vegetation and support desired ecosystem services requires an understanding of the reciprocal linkages between vegetation, hydrology, sedimentation and channel form.

With continued dam building and retrofitting and expansion of existing impoundments, comes opportunity for managing flow regimes to better balance human and ecosystem needs for water. Environmental flow management is one tool that has the potential to restore processes and functioning of riparian ecosystems through strategically managing flows at appropriate times and quantities to optimize yield on the investment (Arthington and Bunn 2006). Through strategically managing the timing, frequency, magnitude, duration, interannual variability flows along rivers, human water needs can be met while supporting river functions. The potential for incorporating environmental flows into river management is great. For example, in the past decade, the US Forest Service has participated in over 100 FERC hydropower

relicense proceedings. These proceedings provide opportunities to incorporate biotic considerations into flow management plans downstream from hydropower facilities on public lands and to retrofit or re-operate facilities to provide such flows. Cooperative relationships between federal agencies and NGOs can also provide opportunities for balancing conservation with human demands for water (Richter et al. 2006). For example, the Sustainable Rivers Project (SRP) is a collaboration between the Nature Conservancy and the US Army Corps of Engineers aimed at incorporating river conservation into dam management. Utilizing the concept of ecologically sustainable water management (ESWM; *sensu* Richter et al. 2003, 2006), the aim of the seven projects currently underway is to meet human demands for water while using dam operations to restore and protect the health of rivers. Such management can provide the template for restoration and/or maintenance of riparian ecosystems over a much larger spatial extent than is possible through active, site-specific management of stream channels and vegetation.

Through strategically managing the timing, frequency, magnitude, duration, interannual variability of flows to accommodate desired processes along river, human water needs can be met while supporting river functions.

Proper management of vegetation cover and soil stability (e.g., livestock, agriculture, urbanization, roads, fire, mining, timber extraction, etc.) throughout the watershed is increasingly important for riparian ecosystems systems that may already be stressed due to water extraction or flow alteration. However, management of riparian areas themselves is paramount as healthy riparian ecosystems serve as buffers between upland activities and aquatic ecosystems and healthy riparian ecosystems are more resistant and resilient to perturbations and external stressors or environmental threats.Through fire and fuels management, livestock grazing during seasons and in places that minimize impacts to the channel (e.g., through placement of watering areas, salt licks, and exclosures), well-designed roads and stream crossings to minimize negative effects on stream channels, well-managed forestry, management of recreational activities such as camping and ORV use, and a range of other management activities, watershed scale and local factors can complement one another in sustaining riparian health.

Proper management of vegetation cover and soil stability throughout the watershed is increasingly important for riparian ecosystem needs that may already be stressed due to water extraction or flow alteration. However, management of riparian areas themselves is paramount as healthy riparian ecosystems serve as a buffer between upland activities and aquatic ecosystems and healthy riparian ecosystems are more resistant and resilient to perturbations and external stressors or environmental threats.

This assessment of threats to riparian ecosystems of the western United States has utilized the best available geospatial information to highlight riparian areas that are most likely to at risk of degradation or further degradation based upon the underlying processes that historically supported and continue to influence these systems. Vulnerability to pathogens and disease, loss of habitat complexity and quality, invasion by non native plants and animals, adverse effects of intense and frequent fires, recreational and livestock impacts, and other threats to riparian condition are all influenced by the fundamental physical processes considered in this threats assessment. Though data of sufficient quality to consider such specific stressors to riparian areas at the scale the western US are not available at this time, we took advantage of existing data to consider the fundamental processes that influence these smaller-scale stressors through examining hydrology, sediment, and the factors that support fluvial processes. Through managing rivers to accommodate the basic processes that support western riparian areas, land owners, managers, and users can provide the template or foundation for healthly systems. Given the potential to support diverse riparian ecosystems, land and resources may be further managed through best management practices and site- and context specific management plans to sustain desired properties and ecosystem services from these most valuable components of western landscapes.

Through manaaging rivers to accommodate the basic processes that support western riparian areas, land owners, managers and users can provice the template or foundation for healthy systems.

Acknowledgements

We thank J. Salo, S. Litschert, and J. Hicke for reviews of earlier drafts. Scientists at the Western Environmental Threats Assessment Center provided helpful review and comments that improved this work. Funding from this project was provided by the Western Environmental Threats Assessment Center, Prineville, OR (D.M. Theobald, D.M. Merritt and J.B. Norman) and the Watershed, Fish, Wildlife, Air and Rare Plants Program of the USDA Forest Service (D.M. Merritt). Thank you to Joyce VanDeWater for assistance with formatting and layout.

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Appendix.

Detailed methodology

Flow modification

We estimated changes in the stream flow regime using a measure *F* we call "flow modification", which is measured as the ratio of the normal storage volume of a dam (acft) to the natural mean annual "virgin" discharge (assuming no dams or other human modifications). We summed the normal storage volume of dams cumulatively downstream, using data from the National Inventory of Dams (NID; USACE 2008). The NID contains dams that are high or significant hazard classification, at least 25 feet high and at least 15 acft, or at least 6 feet high and at least 50 acft (*n=82,642*). We filtered the dams to identify unique reservoirs (*n=67,662*). Our measure of characterizing flow modification is consistent with the literature (see Dynesius and Nilsson 1994, Nilsson et al. 2005). Values of *F=0.0* indicates no flow modification; a value of *F=1.0* indicates that reservoirs are able to store roughly the average annual discharge flowing through a given stream segment. For roughly 20% of watersheds, flow fragmentation values were higher than 1.0, which can occur legitimately when there are large reservoirs able to capture wet years, or when a reservoir receives water from a trans-basin diversion (e.g., the Horsetooth Reservoir receiving water from the Colorado-Big Thompson project on the Colorado Front Range). Very large values (*F>10.0*) can also occur as a result of artifacts in our definition of catchments as well as small tributary watersheds connecting to very large reservoirs. For both cases we capped the *F* value for all catchments to be a maximum value of 10.0, to minimize the effects of scale artifacts.

We estimated mean annual virgin discharge using regression-based equations between watershed attributes and climatic variables developed for 18 regions in the US by Vogel et al. (1999):

$$\tilde{\mu}_Q = e^a A^b \mu_p^{\,o} \mu_T^{\,d}$$

where $\tilde{\mu}_Q$ is the average annual stream discharge (cms), *a* is a power estimated for water resource regions, A^b is the area of catchment (km^2) set to the *b* power, $\mu_p^{\,o}$ is the average precipitation (mm) within catchment set to the *o* , and $\mu_T^{\,d}$ is the average temperature (degrees F * 10) set to the *d* power. Note that the power parameters were specific to 18 water resource regions. For each reach catchment (using USGS NHD Plus at 1:100,000 scale) we calculated this equation to estimate local catchment average annual discharge. We then generated a hydrologic network using the FLoWS tools (Theobald et al. 2006) and then accumulated discharge downstream. For the past scenario, we used precipitation and temperature data for 1900-1940 from PRISM (2006). For the future scenario, we estimated future mean virgin steam flow using temperature and precipitation estimates for IPCC scenarios A1B and B1 that uses the Community Climate System Model v3.0 from the National Center for Atmospheric Research (UCAR 2007).

Sedimentation

We used the Revised Universal Soil Loss Equation (Renard et al. 1997) to estimate the amount of sediment production. RUSLE uses five major factors to compute the average annual erosion or sediment produced in a watershed: rainfall erosivity (R), soil type (K), topography composed of length (L) and slope (S), cover type (C), and management practices (P).

$$A = R \cdot K \cdot (LS) \cdot C \cdot P$$

where *A* is the computed spatial average soil loss and temporal average soil loss per unit area (tons/ha/yr), *R* (Mj mm/(ha h yr)), *K* (tons ha h/(ha MJ mm)), *L* the slope-length factor, *S* the slope steepness factor, *C* the cover management factor, and *P* the conservation support practice factor. To compute RUSLE, we generated a series of raster datasets at 30 m resolution for each of the RUSLE factors.

R Factor

The *R* factor represents the driving force of sheet and rill erosion by rain fall and runoff as a function of rainfall amount and intensity. Due to the large spatial extent of this analysis and the data requirements required to calculate the RUSLE rain fall erosivity factor we relied on a dataset compiled by the EPA (USEPA 2009) that contains a variety of metrics summarized by and 8-digit Hydrologic Unit Code. We converted the HUC 8 features (polygons) to 30 m resolution to match the USGS National Elevation Dataset DEM (Gesch et al. 2002). Note that we used these values also for the past and future conditions – as it is challenging to derive hourly-to-daily rainfall intensity estimates from global climate models that provide precipitation at monthly to yearly time periods.

K Factor

The *K* factor is an empirical measure of soil erodibility as affected by intrinsic soil properties. These soil properties include soil texture, organic matter, structure and permeability of the soil profile. The K values used for this study were derived from the Natural Resource Conservation Service (NRCS) State Soil geographic (STATSGO) database (http://soils.usda.gov/survey/geography/statsgo/) and they are expressed as annual averages in English units converted to SI metric units according to Foster et al. (1981).

L & S Factors

The L and S factors in RUSLE reflect the effect of topography on erosion. There are a variety of empirical formats capable of calculating the L and S factors, including McCool et al. (1993) and Desmet & Grovers (1996). We chose to use more recent modifications developed by Winchell et al. (2008) because they build on earlier work to better incorporate upslope area and have tested their GIS approach against field observations reported in the NRCS National Resources Inventory. We calculated the slope *S* factor using the continuous function developed by Nearing (1997), where slope *s* is measured as percent slope (not degrees) and *theta* is measured as slope in radians (Winchell et al. 2008) using ArcGIS methods[1].

$$S = -1.5 + 17/\left[1 + \exp(2.3 - 6.1 \sin\theta)\right] \quad \text{(for } s < 55\%\text{)}$$

L_i is the L-factor for cell *i* and is computed as follows:

$$L(i,j) = \frac{(A(i,j) + D2)^{m+1} - A(i,j)^{m+1}}{x^m \cdot D^{m+2} \cdot (22.13)^m}$$

where $A_{i\text{-}in}$ is the contributing area at the inlet of cell *i* and is measured in m^2; *D* is the cell size in m; x_i is the shape factor based on the aspect (α) of the slope for cell *i*.

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[2]S_rad = slope (DEM, DEGREES) div DEG

A_rad = aspect (DEM) div DEG

Sf = -1.5 + 17 / (1 + exp (2.3 - 6.1 * sin (max(S_rad, 55.0 div DEG)))

$$x_i = \sin \alpha_i + \cos \alpha_i$$

$$\beta = (\sin\Theta/0.0896)/ \left(3.0(\sin\Theta)^{0.8} + 0.56\right)$$

These are implemented as a series of ArcGIS map algebra statements. Note that we used the NHD Plus flow accumulation raster to compute the slope length, which is based on a eight flow direction (D8) on a "filled" 30 m DEM. Winchell et al. (2008) found this method to be less prone to extreme slope values because it considers a broader landscape slope (than other methods such as D*inf* by Tarboton 1997). We used a threshold of 1,000 cells (30 m) to distinguish streams from overland flows after comparing synthetic streams to 1:24,000 NHD "blue-lines" (Burnett et al. 2007).

C Factor

The cover factor *C* is a weighting scheme that reflects the effect of land use activities and land cover on erosion rates. This factor accounts for how a land use activity or land cover shelters soil from rain and surface runoff. The *C* factor parameterization for the three scenarios (natural, current, and future) was derived from various source tables in different documents related to RUSLE. The past scenario parameterization involved a lookup table that generalized the 473 land cover classes from the Biophysical Settings (BpS) types from LANDFIRE (Rollins and Frame 2006; www.landfire.org) into 16 general vegetation classes and assigned a *C* factor (Table 3). We generated the C factor raster for the current and future scenarios by integrating a variety of different land cover datasets, including the Existing Vegetation Types (EVT) from LANDFIRE, low density housing for 2000 and 2030 from SERGoM (Theobald 2005), and roads from the ESRI Streetmap 2006 (ESRI 2009). The EVT dataset was assigned C factors using the same general classes used for the BpS land cover dataset with human modified classes assigned a C factor (Table A1).

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[3] beta = ((sin(S_rad) / 0.0896) / (3 * pow(sin(S_rad), 0.8) + 0.56))

m = beta / (1 + beta)

x = abs(sin (A_rad) + cos (A_rad))

fac = flowaccumulation… on overland slopes, not streams

LS = max(sf * (pow((fac * 900) + 900, m + 1.0) – pow((fac * 900), m + 1.0)) / pow (30, m + 2.0) * pow (x * 22.13, m)), 20.0)

Table A1. C factors for past or “natural” scenario of RUSLE models.

	Land cover	C Factor	Citation
Natural types	Barren	1.0000	Toy and Foster 1998
	Coniferous Forest	0.0020	Breiby 2006
	Deciduous Forest	0.0010	Breiby 2006
	Deciduous Shrubland	0.0250	Breiby 2006
	Dense Grassland	0.0800	Dawen et al. 2003
	Floodplain Forest	0.0100	Breiby 2006
	Lowland Coniferous Forest	0.0025	Breiby 2006
	Lowland Deciduous Forest	0.0015	Breiby 2006
	Marsh/Riparian/Wetland	0.0010	Breiby 2006
	Medium-tall grassland	0.0120	Breiby 2006
	Mixed Forest	0.0010	Breiby 2006
	Mixed Forest woodland	0.0020	Breiby 2006
	Open Water/Exposed Rock	0.0000	Breiby 2006; McCuen 1998
	Shrubland Other	0.0290	McQuen 1998
	Snow field	0.0010	Dawen et al. 2003
	Sparse Grassland	0.2000	Dawen et al. 2003
Human-modified	Aggregate mining	1.0000	Guobin et al. 2006
	Asphalt	0.0001	Toy and Foster, 1998
	Cultivated Crops Irrigated	0.2400	McCuen, 1998
	Developed General	0.0030	Guobin et al. 2006
	Developed Suburban	0.0020	Guobin et al. 2006
	Developed Urban	0.0010	Guobin et al. 2006
	Fallow	1.0000	McCuen, 1998
	General Cropland	0.5000	Dawen et al., 2003
	Gravel	0.2000	Toy and Foster, 1998
	Industrial	0.0050	Guobin et al. 2006
	Mixed Urban	0.0040	Guobin et al. 2006
	Paddy field	0.1000	Dawen et al., 2003
	Pasture Hay	0.1400	McCuen, 1998
	Recreational Grasses	0.0080	McCuen, 1998
	Small Grains	0.2300	McCuen, 1998

The *C* factor designation for roads consisted of delineating different road types from the ESRI streetmap 2006 dataset via the CLASS_RTE attribute and assigning area of influence (meters) and proportion of pervious and impervious surface values (Table A2). This method developed by Toy and Foster (1998) incorporates road surface, vegetation, and bare soil to a road cell. For example, a U.S. highway has a 60 m (width) area of influence and is made up of 75% asphalt (C = 0.0001), 12.5% dense grassland (C = 0.035) and 12.5% bare soil (C = 1.0) with an average C factor of 0.13. This compared to a gravel county road that has an area of influence of 30 m and a *C* value of 0.7.

Table A2. RUSLE C and P factors for road classes.

Road Type	Total Foot print (meters)	% Impervious	% Grassland	% Bare Soil	C Factor	P Factor
U.S. Interstates	90	75	12.5	12.5	0.13	0.75
U.S. Highways	60	75	12.5	12.5	0.13	0.70
Paved on ramps	30	50	25	25.0	0.25	0.80
Paved County roads	30	75	10	15.0	0.16	0.80
Paved Urban streets	30	75	25	0	0.02	0.80
Gravel County roads	30	0	10	90.0	0.70	1.00

The EVT dataset supplies current vegetation and some land use classes (development intensities, mining, and agriculture) providing the platform for the current and future scenarios. Land use datasets used to parameterize current and future cover and management practices utilized the ESRI Streetmap 2006 and SERGoM housing density (Theobald 2005) for 2000 and 2030. The SERGoM housing density surfaces for the years 2000 and 2030 were used to weight EVT C factors where vegetation and agricultural cover types where intermixed with suburban and exurban development. This is necessary for two reasons: First, the EVT land cover developed classes (20, 21, 22, 23 and 24) only capture housing densities at urban levels. The designation of C factors for the SERGoM dataset began by breaking the continuous housing density values into three discrete groups urban (< 0.1 ha per housing unit), suburban (0.1 – 0.68 ha per housing unit), and exurban (0.68 – 16.18 ha per housing unit) as defined by Theobald (2005). Housing densities > 16.8 ha per housing unit were not evaluated due to lack of C factor data. The urban and suburban classes were assigned a C factor (Table A2) and averaged with the C factors defined for the EVT classes to account for urban and suburban areas within agricultural and natural land cover classes. The final C factor surface for 2000 EVT was developed by burning in the roads C factors on top of the SERGoM informed EVT C factor surface.

For the scenario reflecting 2030, we altered the land cover conditions using existing roads, EVT, and developed lands for 2030. The SERGoM 2030 provides information on how urban, suburban and exurban areas will expand spatially thus altering sedimentation rates and distribution. To summarize, the natural scenario represents landscapes that have not been modified by humans being composed of native plants, the current scenario represents a landscape that has been modified by humans with alterations to vegetation type and distributions as well as disturbance effects, and the future scenario represents current vegetation, road, and agriculture conditions modified to reflect probable development in 2030.

P Factor

The P factor accounts for control practices that reduce the erosion potential of runoff by their influence on drainage patterns, runoff concentration, runoff velocity and hydraulic forces exerted by runoff on soil (Renard et al. 1997). Human influences on soil erosion control are important to include in the P factor, but there is no western US or statewide reference because erosion control is a very local activity. In this study, a P factor is assigned to human modified land cover classes only (e.g., Agriculture, Developed, and Roads) based on P factors from multiple sources. P factors for road and SERGoM classes were assigned using the methods used to define C factors for roads and SERGoM classes (Tables A1, A2, and A3). The final P factor surface for 2000 and 2030 EVT was developed by burning the roads P factors on top of the SERGoM informed EVT P factor surface.

Table A3. RUSLE P factors for human modified land cover classes

Landcover	P Factor	Citation
Aggregate mining	1	Guobin et al. 2006
Asphalt	0.0001	Toy and Foster,1998
Cultivated Crops Irrigated	0.35	McCuen, 1998
Developed General	0.001	Guobin et al. 2006
Developed Suburban	0.001	Guobin et al. 2006
Developed Urban	0.001	Guobin et al. 2006
Fallow	0.85	McCuen, 1998
General Cropland	0.35	Dawen et al., 2003
Gravel	1	Toy and Foster, 1998
Industrial	0.001	Guobin et al. 2006
Mixed Urban	0.001	Guobin et al. 2006
Paddy field	0.85	Dawen et al., 2003
Pasture Hay	0.85	McCuen, 1998
Recreational Grasses	0.04	McCuen, 1998
Small Grains	0.85	McCuen, 1998

We multiplied the factors together on a cell by cell basis, which results in the estimated sediment produced at each cell. We then summarized by reach catchment to compute the estimated total sediment within a catchment, and aggregations up to a HUC8 and WRR. Note that cells defined as a stream did not generate any sediment. Also note that we are estimating sediment production here, but are not explicitly representing the transport of that sediment down to the nearest stream, as this is a very challenging effort with few empirical studies to rely on to parameterize our model (but see Miller and Burnett 2008). To summarize the amount of sediment produced by reach catchment areas, we used Zonal statistics (sum) to calculate total sediment load that influences stream reaches, resulting in an estimated average soil loss per unit area (tons/ha yr). This was executed for each of the three scenarios, and then the percent difference was calculated to evaluate relative change in sediment loads.

Riparian zone – lateral connectivity

To estimate the direct loss of riparian zone areas, as well as the loss of lateral connectivity, we measured the proportion of human-dominated land uses inside the valley bottom (Figure A-1). Human-dominated land cover types within the riparian zone were estimated based on reclassified urban and agricultural cover types from LANDFIRE to be 1.0 and all other “natural” types 0.0. We also included other areas in the riparian zone that had either major roads (highways, secondary roads) or at least exurban (1 unit per 16 ha) or higher housing density. Note that this does not include loss of riparian zones due to inundation from reservoirs.
Valley bottoms have been modeled a number of ways (e.g., Williams et al. 2000; Clarke et al. 2009), and we modeled valley bottoms by growing allocation zones away from streams (USGS NHD Plus 1:100,000) in a 30 m elevation raster (USGS National Elevation Dataset). We used the index of valley constraint (V_c; Burnett et al. 2007) where $V_c = VFW/ACW$. VFW is the valley floor width that is estimated as the length of a transect that intersects the valley walls at 2.5 times the estimated bank-full depth (H_{bf}). We computed $H_{bf} = 0.36A^{0.2}$, where A is drainage area (km^2). The active channel width is $ACW = 2.19108 + 1.32366 * D^{0.5}$, where D is the mean annual discharge (ft^3/s).

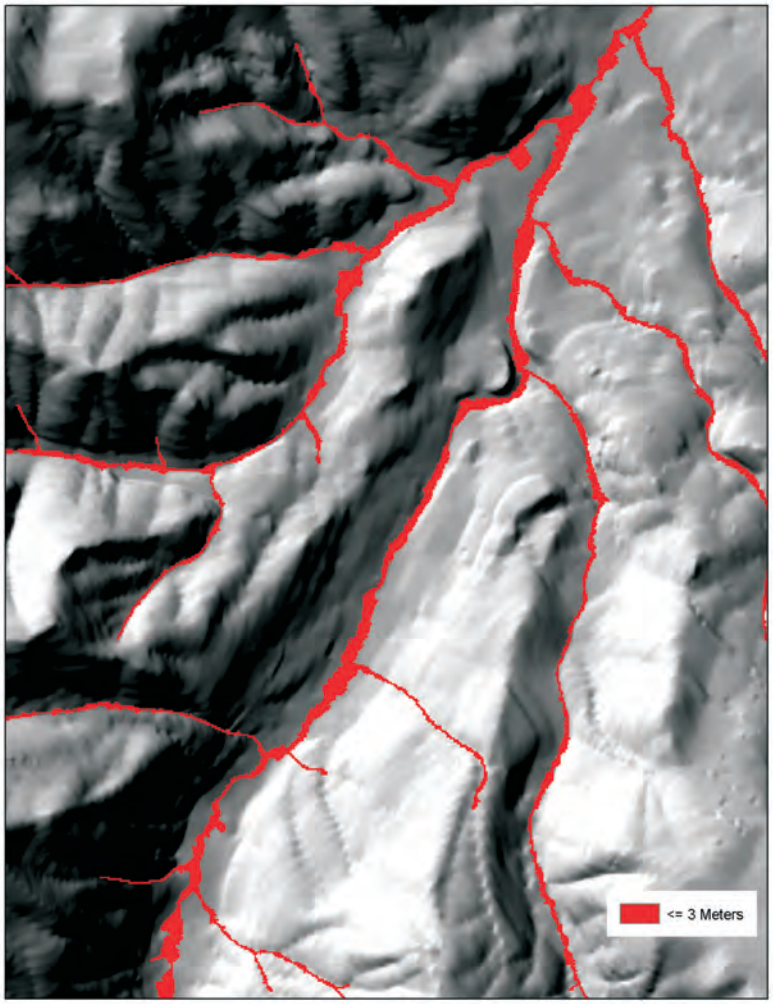

Figure A-1. An example of modeled valley bottom/potential riparian zone computed at 30 m using National Hydrography Dataset Plus 1:100,000 and National Elevation Dataset.

Synthesis of process factors into a riparian condition score

Table 4 provides a summary of the three process factors and datasets used. We also standardized the raw data from the individual factors to compute a composite index that could be used to evaluate the relative "most" from the "least" affected watersheds. To do this for flow fragmentation, we standardized the current (2006) raw values using the mean and standard deviation computed from the historic (1940) values. Similarly, we standardized the flow fragmentation for future scenarios using the mean and standard deviation of the current (2006) values. This standardization resulted in values that range (roughly) from -1 to +1, where negative values denote a reduction in flow fragmentation and positive values denote an increase in fragmentation (i.e., a least effected). By standardizing using an earlier time frame, we are able to compare departure from previous conditions. We standardized the sediment factor values by calculating the mean and standard deviation of the historical sediment production values (based on Biophysical Settings). Similarly, we standardized future scenarios of sediment production by finding the mean and standard deviation of the current scenario values, but we computed and absolute value of the standardized values. That is, we assumed that any change – whether an increase or decrease in sediment production resulted in a positive value denoting an increase in sediment produced (i.e. least effected). The riparian zone factor was converted into a 0.0 to 1.0 index by simply using the proportion of human land use in the potential riparian zone as compared to historical (no human land use).