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A Multi-scale Spatial Approach to Address Environmental Effects of Small Hydropower Development

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Abstract Hydropower development continues to grow worldwide in developed and developing countries. While the ecological and physical responses to dam construction have been well documented, translating this information into planning for hydropower development is extremely difficult. Very few studies have conducted environmental assessments to guide site-specific or widespread hydropower development. Herein, we propose a spatial approach for estimating environmental effects of hydropower development at multiple scales, as opposed to individual site-by-site assessments (e.g., environmental impact assessment). Because the complex, process-driven effects of future hydropower development may be uncertain or, at best, limited by available information, we invested considerable effort in describing novel approaches to represent environmental concerns using spatial data and in developing the spatial footprint of hydropower infrastructure. We then use two case studies in the US, one at the scale of the conterminous US and another within two adjoining rivers basins, to examine how environmental concerns can be identified and related to areas of varying energy capacity. We use combinations of reserve-design planning and multi-metric ranking to visualize tradeoffs among environmental concerns and potential energy capacity. Spatial frameworks, like the one presented, are not meant to replace more in-depth environmental assessments, but to identify information gaps and measure the sustainability of multi-development scenarios as to inform policy decisions at the basin or national level. Most importantly, the approach should foster discussions among environmental scientists and stakeholders regarding solutions to optimize energy development and environmental sustainability.

Keywords Dams · Energy policy · Reserve design · Marxan · Landscape ecology

Introduction

Hydropower development continues to grow worldwide to provide a means of energy expansion in developed and developing countries (Grumbine and Pandit 2013; Zimny et al. 2013). The construction of large hydropower facilities is on the forefront of the world's largest environmental debates, with the most publicized construction occurring in underdeveloped countries, such as the Mekong Basin in China (Huang and Yan 2009; Ziv et al. 2012), the Amazon in Brazil (da Silva Soitoa and Freitas 2011), and the Himalayan Region in India (Grumbine and Pandit 2013). To a lesser extent, small hydropower development continues to grow as a means to provide an energy source in response to demands from local economies in developed (Balat 2007; Punys and Pelikan 2007; Malesios and Arabatzis 2010; Yuksel 2010; Alonso-Tristán et al. 2011) and undeveloped countries (Anderson et al. 2006; Ohunakin et al. 2011). Although the scale of small hydropower development is presumed to have lesser or benign environmental effects than the construction of large facilities, the cumulative environmental consequences of small versus large development are still equivocal (Chin et al. 2008; Kibler and Tullos 2013; Jager and McManamay 2014).

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With the growth of hydropower, many frameworks were developed to evaluate the feasibility of construction costs relative to energy gained (McNally et al. 2008; Mishra et al. 2011; Punys et al. 2011; Sharma et al. 2013). Indeed, many resources are available for siting locations suitable for hydropower development based solely on high energy potential (Larentis et al. 2010; Yi et al. 2010; Punys et al. 2011). In terms of environmental assessments, multiple frameworks are available to conceptualize ecological and physical responses to dam construction at the site-specific scale (Brandt 2000; Burke et al. 2009) or organize on-the-ground environmental assessments of potential new site locations (AMEC 2011; VANR 2008). Less common, however, are widely applicable and nationally transferable frameworks that measure the potential environmental effects of hydropower development (Brown et al. 2009; Burke et al. 2009; IHA 2010; MRC 2010). One example is the Integrative Dam Assessment Modeling (IDAM) tool, which was developed to assess the relative cost–benefits of proposed dam construction by evaluating biophysical, socio-economic, and geopolitical perspectives into a decision-supported framework (Brown et al. 2009). Similarly, the International Hydropower Association developed a Hydropower Sustainability Assessment Protocol (HSAP) as a ranking based system that considers the political, socio-economic, and environmental sustainability according to whether best practices are in place for existing or future projects (IHA 2010).

While information-rich investigations of individual hydropower sites are feasible, these approaches are unrealistic for widespread competing hydropower development at national, or even regional, scales (Elerwein 2013). Recent evidence suggests that conventional approaches to environmental assessments, such as EIAs, are challenged to keep up with the rapid expansion of hydropower (Elerwein 2013). Even more troublesome, EIAs fail to address more comprehensive issues of hydropower development at the basin or regional level because they are constrained to the individual project level (Tullos 2009; Elerwein 2013). In contrast, more holistic applications are needed that can be used alongside frameworks locating areas of energy potential while also addressing widespread concerns, identifying areas of similar environmental issues, and prioritizing areas for development (de Almeida et al. 2005). These applications could be used as a precursor to more detailed analyses, such as EIAs. In addition, project costs can be reduced when the environmental issues are addressed early in the planning phases rather than later or even worse, as an afterthought.

Spatial applications, i.e., frameworks with the explicit purpose of using patterns in spatial data to represent environmental processes, have been under-utilized to document the consequences of hydropower development in hindsight, either at the project level (e.g., Pathak 2008; Dukiya 2013; Zhao et al. 2013) or for widespread development (e.g., Lehner et al. 2011). Even more so, such approaches are rarely used to plan for large-scale hydropower development, taking into consideration multiple projects (e.g., Rojanamon et al. 2009; Pandit and Grumbine 2012; Kibler and Tullos 2013). We define large-scale hydropower development as cases where multiple cumulative or competing projects are considered simultaneously as to increase basin- or national-level energy portfolios. As one example, Kibler and Tullos (2013) developed 14 spatially affiliated metrics to assess the alternative impacts of small versus large hydropower development proposed for the Nu River Basin, China. While their study is among the most holistic landscape-based approaches to assessing environmental impacts in the peer-reviewed literature, the authors readily admit that their analysis was limited by available information and the results were unique to the setting (Nu River Basin, China). Two other studies used geospatial analyses to address environmental concerns to potential large-scale hydropower development in Thailand and Vietnam but did not fully document the applied methodologies (see Rojanamon et al. 2009; WB 2009, respectively). The lack of documented methods to support large-scale hydropower planning is problematic when countries face the need to predict environmental concerns at large spatial scales as to inform national energy budget assessments. The paucity of approaches for landscape analysis related to hydropower planning is likely due to the fact that the majority of hydropower construction in developed countries preceded the use and availability of landscape planning tools, such as GIS technologies. Furthermore, construction of facilities globally, and any associated environmental impact assessments, typically occurs on a singular project-by-project basis (Elerwein 2013), rather than considering many facilities simultaneously.

Given the rapid growth in interest in renewable energies worldwide, approaches are needed that provide spatial evaluations of large-scale energy development. Herein, we propose a spatial approach for assessing environmental impacts to hydropower development at large spatial scales. The spatial approach is not meant to override or replace more in-depth environmental assessments, but can provide an initial reconnaissance-level assessment of the environmental context, identify information gaps, and design environmental impact studies. Specifically, the purpose of the spatial approach is to compile and organize environmental data as to facilitate stakeholder involvement in assessing potential environmental concerns relative to potential energy gained. The entire approach depends largely on stakeholder involvement including fundamental factors, such as hypothesizing what are environmental concerns, and more complex needs, such as ranking or prioritizing environmental concerns. Most importantly, the approach fosters discussions of approaches to sustainable energy development as to inform policy decisions at the scale of basins, regions, or entire nations.

Multi-scale Approach Overview

For brevity, the approach we present primarily focuses on assessing potential ecological and biophysical effects to hydropower development; however, we include some factors (e.g., water use, recreation) that are highly inter-related. Hence, the approach can be expanded to include socio-economic and geopolitical factors. We devote considerable time to developing a conceptual framework of hydropower-related environmental effects; however, concepts never mature if unsupported by data. In many countries, hydropower planning assessments are limited by available information; thus, our approach is data-centric in that we expend a great deal of effort in describing novel approaches to represent environmental concerns using available information, geospatial analysis, and model building as to not compromise high granularity with increasing scale. The spatial approach includes five steps, each guided by social roundtable discussion: (1) hypothesis generation, (2) defining data needs to address hypotheses, (3) developing novel ways to compile and create data, (4) determining the spatial footprint of hydropower, and (5) ranking and prioritizing development (Fig. 1).

Hypothesis Development

Hydropower development modifies terrestrial and aquatic ecosystems thereby impacting a multitude of ecological and biophysical processes and feedback mechanisms (see Burke et al. 2009 for a review). Because defining environmental effects is a matter of scale and the choice of the observer (Allen and Starr 1982), identifying the spatial extent and resolution of the analysis is typically needed first. We considered environmental effects at the scale of hundreds of meters to entire basins because this spatial resolution was the most appropriate when considering the development of entire hydropower projects (reservoirs, dams, and downstream environments). After identifying the appropriate scale, developing hypotheses, based on rigorous scientific literature review, can help avoid subjectivity in defining what environmental variables and related spatial and temporal resolutions in include. Hypothesis generation aids in guiding a geospatial approach to large-scale applications rather than developing an exhaustive list of all issues, some of which can only be addressed through on-the-ground analysis. EIAs have been used for decades to inform the process of hydropower development (Tullos 2009). While EIAs provide a robust foundation for identifying potential effects and uncertainties, EIAs tend to become extremely complicated as to encompass all possible ecosystem responses and pathways (Bruns et al. 1993; Tullos 2009). Not surprisingly, EIAs can take months to prepare (Elerwein 2013) and this level of detail is not feasible for widespread development. Equally problematic, the current US site-by-site regulatory regime can create an extremely narrow view of issues that can only be identified through site visitation. The challenge here is focusing on hypotheses that can be addressed through geospatial analysis at large-scales and prioritizing the data needed to conduct the analysis.

Define Data Needs

Data searches can become disorganized and expensive unless structured around specific pre-defined hypotheses (Fig. 1). Hypotheses clarify the level of detail, specifically the scope, breadth, and resolution of spatial coverage, required to address environmental effects (O'Neill et al. 1986). Because capturing process-driven impacts in large-scale assessments is difficult, the tendency might be to limit analyses to only effects that can be easily captured with existing data. Spatial applications must present a compromise between only utilizing existing data, thereby limiting conclusions, and resource-intensive modeling whose results are only applicable at the site-specific level.

Differentiating between direct versus indirect effects can aid in determining what data might be required. Direct effects do not involve an intermediate step in the path from development to the environmental response. As an example, a planned reservoir may overlap with sensitive lands. We can presume there is some direct compromise of the ecological value the lands offer if construction ensues. In contrast, effects of reservoir construction and operation on downstream aquatic communities may take years to attain, especially if mediated by hydrologic or water quality factors (Quinn and Kwak 2003). Predicting the full suite of environmental effects of hydropower development (especially those that are process-driven) using coarse spatial applications is unpractical. Hence, differentiating between causal and preventative approaches to hydropower planning can also be used to prioritize data needs. Causal approaches rely on isolating response pathways in order to predict the magnitude and direction of environmental impacts. For example, predicting the effects of dam construction on water quality is uncertain unless the reservoir storage, residence time, intake depth, operational regime, physiochemical setting, and land-use are known and predictive models are available. In contrast, preventative approaches consider potential environmental impacts, but

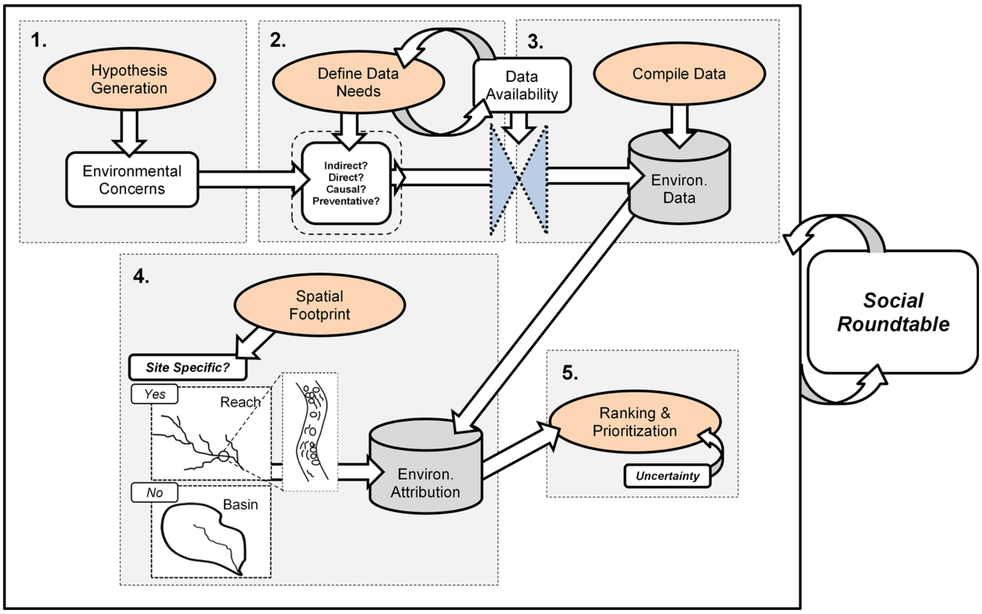


Fig. 1 Conceptual model outlining five steps of the landscape-based approach to address environmental effects of hydropower development. (*1*) Hypothesis generation is used to identify relevant environmental effects. (*2*) Data needs are defined to address hypotheses. (*3*) Novel methods are used to compile or create data based on availability of information. (*4*) Spatial interactions (i.e., spatial footprint) are used to define how environmental variables are summarized. (*5*) Potential sites or basins are ranking and prioritizing following an incorporation of uncertainty

acknowledge the full response pathway is unknown. However, preventative approaches also take into consideration the existing environmental condition of the landscape. For example, the environmental effects of hydropower development may be more or less benign depending on the environmental context. Potential sites for hydropower development may vary in their existing anthropogenic-induced disturbances and in their importance in serving as biological reserves within the river basin (Pringle 2001). As an example, a preventative approach may rely solely on reservoir size or storage as a surrogate for potential downstream ecological and biophysical effects, since reservoir size approximates the propensity for potential habitat changes (e.g., Kibler and Tullos 2013). A slightly more complex preventative approach may include characterizing reservoir size (e.g., degree of regulation) in conjunction with spatial proximity to water quality concerns, which can then be used to assess the potential compounding or counteracting effects (i.e., potentially restorative) of hydropower development on water quality. Identifying these key elements will help clarify what data are needed to answer specific questions.

Environmental Data Compilation

Once data needs are diagnosed, intensive data drives are conducted through internet-based open-access repositories or by contacting agencies that house information to obtain existing environmental data. However, the data required to adequately address hypotheses are not synonymous with scope, scale, and resolution of available information. Countries, local governments, and agencies vary considerably in curating, maintaining, and disseminating geographic information. Hypothesis generation and to some extent, data availability, will filter how and what environmental concerns are addressed in hydropower planning (Fig. 1). In addition, much data may require synthesis or modeling, such as extrapolating species presence and absence to unsampled areas.

Spatial Footprint

Determining potential environmental effects related to hydropower development is complicated by undefined interactions between a given stressor and its impact. Dams vary considerably in their lateral, vertical, and longitudinal biophysical effects (Stanford and Ward 2001), which depend upon the climatic, geomorphic, political, regulatory, and social context. Likewise, the spatial footprint of hydropower development is highly variable. Individual reservoirs can range from <0.1 to >5.4 $\times$ 10^5 ha (Rosenberg et al. 1997); however, the full extent of longitudinal effects may extent for hundreds of miles downstream (Ellis and Jones 2013). According to Stanford and Ward (2001), the downstream longitudinal effects of dam regulation varied from 0 km in the Loire River in France to over

472 km in the Colorado River, as measured by biotic or physical processes offset by dam regulation. However, Pringle (1997) suggested that potential disturbances could not only be transmitted in the downstream direction, but also to areas upstream of reservoirs. In addition to longitudinal effects, estimates of lateral effects range from 1,000 to 6,000 m, depending on the environmental variable (Bohlen and Lewis 2009; Rojanamon et al. 2009; Soussan et al. 2009; Zhao et al. 2013).

Defining spatial overlap will determine how geospatial environmental data relate to locations of hydropower development and also the level of geoprocessing required to assemble environmental information (i.e., environmental attribution database) (Fig. 1). If countries are attempting to increase their overall energy budget, basin-level analyses can be used for reconnaissance-level planning or to organize general regional concerns. In contrast, site-specific assessments require establishing a spatial footprint of hydropower development that varies depending on the size of the project and the environmental variable of interest (Fig. 1). The spatial footprint should include areas of impact, such as the area inundated by the reservoir, area of construction surrounding the dam, and the estimated downstream extent of impacts. Tributary inputs, more than any other factor, ameliorate the downstream effects of an impoundment (Ward and Stanford 1983; Stanford et al. 1994; Vinson 2001; Ellis and Jones 2013). Thus, the downstream extent of dam construction could be demarcated on the basis of tributary junctions as well as dam size.

Once the spatial footprint has been defined, determining potential environmental effects on the basis of spatial overlap, rather than spatial proximity, may be misleading. For example, Kibler and Tullos (2013) suggested that land conversion may occur in areas adjacent to, rather than overlapping, locations of dam construction. Because the spatial extent of hydropower impacts on ecological or biophysical processes in the landscape is likely unknown, incorporating variable buffers into analyses can provide an estimate of distances suitable for detecting environmental effects (Zhao et al. 2013) while also providing an estimate of uncertainty of environmental effects with distance. Buffer analysis using concentric rings has been used to suggest appropriate riparian buffer widths (Xiang 1996) or determine spatial effects of road construction (Liu et al. 2008) and urban expansion (Li et al. 2010), but has rarely been used to assess the spatial impacts of hydropower construction (Zhao et al. 2010, 2013).

Ranking and Prioritization

Determining associated costs of assessing and addressing environmental concerns, via required or voluntary mitigation, can be an effective way to conduct cost/benefit analyses of energy development. However, this assumes mitigation requirements and associated costs are known. Given the limited information, heuristic assessments can be a simpler alternative and fit well within preventative-type approaches. Prioritizing and preserving areas of conservation value that have little disturbances (i.e., reserves) can be assessed at the expense of various reserve-design costs (McDonnell et al. 2002; Ball et al. 2009), which in this case, can be reflected by losses in potential energy. An example of this approach includes GAP analyses, i.e., identifying gaps in existing conservation efforts, such as areas unprotected from development, and prioritizing where to fill gaps with management resources (Sowa et al. 2007). A commonality in GAP approaches is isolating and prioritizing land acquisition in areas that support vulnerable species, have low environmental risks (e.g., not fragmented by roads or development), but lack protection (McPherson et al. 2008). In turn, we can presume that areas characterized by poor environmental conditions and unprotected for conservation purposes may have lesser environmental impacts from development, unless occupied by vulnerable species, thereby adding insult to injury.

Alternatively, environmental stakeholders and managers may prefer scoring or ranking to assess environmental effects using cumulative indices, as these provide flexibility for incorporating user inputs, such as weights or variable selection, in spreadsheet-type programs. Cumulative scores have been used to compare and prioritize alternative hydropower projects by their energy relative to environmental concerns (Rojanamon et al. 2009; WB 2009). The IDAM tool uses subjective measures to produce an overall cost and benefit associated with dam construction (Brown et al. 2009). In addition, the IDAM tool also provides conceptual diagrams to assess the contribution of three main components (biophysical, geopolitical, and socio-economic implications) to overall cost and benefit scores (Brown et al. 2009). Likewise, international communities have adopted the Hydropower Sustainability Assessment Protocol (HSAP), which uses diagrams to illustrate the political, socio-economic, and environmental sustainability for existing or future projects (IHA 2010). These tools provide transparency in the calculation of cumulative scoring criteria by displaying individual components, but also provide the added flexibility of incorporating user-specified criteria.

Social Roundtable

The benefit of the spatial multi-scale approach is that discussions regarding energy development can move beyond the site-level and into discussions of sustainable basin development. An important consideration is that the social roundtable is omnipresent throughout the steps of framework. Stakeholder involvement is an iterative process where workshops can aid in developing or refining hypotheses, identifying important datasets or variables to include, and rank or prioritize new sites. Comparisons of energy and environmental concerns provide an interpretable means to foster discussions among policy-makers and multiple stakeholder groups regarding planning for sustainable hydropower development. The results of these assessments are also transferable in that energy–environmental relationships can be compared at large-scales among different energy development scenarios. Assessments at the national scale can inform policy decisions, influence leverage for development within different energy sectors, aid in prioritizing basins for future research, and inform long-term cost modeling measuring the economic sustainability of future hydropower growth. In contrast, site-level analyses can be used to initiate discussions among multiple local stakeholder groups and prioritize sites for more in-depth environmental impact assessments.

Case Studies: National and Local-Scale Assessments

To provide an illustration of the multi-scale approach, we apply our framework to a recent assessment of hydropower energy potential for the conterminous US. In response to growing US energy demands, Oak Ridge National Laboratory (ORNL) completed a New Stream-reach Development Resource Assessment (NSD) as an analysis of new small hydropower development for the Department of Energy Water Power Program (Kao et al. 2014). The NSD assessment used a geographic approach to determine the potential for hydropower development in stream segments that currently lack hydropower facilities. Based on recent advancements in high-resolution topography and hydrologic datasets, the assessment is likely the most rigorous evaluation of US hydropower potential to date. The assessment provided potential locations of small hydropower development, which were defined as reservoirs not inundating lands above the 100-year Federal Emergency Management Agency (FEMA) and not compromising existing dam infrastructure (Hadjerioua et al. 2013). Although a diversion scenario was considered in the analysis, the results are based upon the dam and powerhouse being integral. Locations of dams and reservoirs were accompanied by estimates of dam height, reservoir volume, reservoir surface acreage, residence time, and energy capacity (MW).

We assess environmental effects related to energy gained at both the national and local scale. For the national scale, we used 8-digit hydrologic unit sub-basins (HUC08) across the conterminous US as a spatial unit for summarizing environmental effects; whereas for the local scale, we focused on two adjacent basins, the Appalachicola–Chattahoochee–Flint (ACF) and Alabama–Coosa–Tallapoosa (ACT) basins, to provide an example of a site-by-site assessment. Within the ACF and ACT, 390 stream reaches were identified as areas currently lacking hydropower facilities. Of these locations, the average potential dam height was 6.1 m (SD = 2.3 m) and ranged from 1.8 to 17.2 m. The associated energy capacity ranged from 0.2 to 32.49 MW and averaged 1.37 MW (SD = 3.32). Many analyses in the ACF and ACT basins used environmental information at the site-level; however, the coarsest resolution used was either the National Hydrography Dataset (NHD) catchments (1:100 k scale) or 12-digit hydrologic unit sub-basins (HUC12). Although socio-economic and geopolotical are eminent with any size of hydropower development, we purposefully consolidate our analysis to ecological and biophysical effects in order to adequately address those topics.

Steps 1–3: Generating Hypotheses, Defining Data Needs, and Compiling Environmental Data

In order to identify major environmental concern themes for the US, we reviewed current scientific literature, Environmental Impact Statement reports, and FERC license approval articles for hydropower projects. We identified six broad categories and twelve sub-categories to organize hypotheses and provide justification for each in Table 1. These categories reflect predominant environmental responses and major regulatory policies related to hydropower development within the US; thus, we do not consider the list exhaustive or applicable worldwide. Internet searches were conducted through multiple websites to gather available information (Supplementary Material 1). Following an initial compilation of data, a panel of 17 reviewers representing multiple federal and state agencies and non-profit-organizations met for a 2-day workshop, evaluated hypotheses, the data sources, and suggested other potential environmental concerns and associated data. All datasets and their sources are listed in Supplementary Material 1. Additional details regarding methods for creating or compiling datasets are provided in Supplementary Material 2.

Biodiversity

Losses in biodiversity have increasingly been documented to result from dam construction (Anderson et al. 2006; Pandit and Grumbine 2012; Ziv et al. 2012); thus, understanding the current status, distribution, and threats to regional biodiversity is essential to support conservation actions and inform development decisions (Jelks et al. 2008). However, only focusing on species recognized by

Table 1 Hypothesized environmental effects within each category and the associated data required to address each

Environmental concerns	Hypothesis	Sources	Data Needs
Biodiversity			
Species of concern	Species falling under protection or ranking for conservation value will be more vulnerable to dam development	Jelks et al. (2008), Ziv et al. (2012), Grumbine and Pandit (2013)	Areas designated as critical to species survival. Species distributions
Vulnerable species	Ecological or life history characteristics (traits) may predispose species to decline following dam development	Winemiller and Rose (1992), Olden et al. (2006), Mims and Olden (2013)	Species distributions. Trait information
Habitat alteration			
Stream channel habitat	Dams reduce sediment and bedload transport coarsening downstream reaches. While the size of the facility will determine the extent of changes, stream characteristics (e.g., slope, sinuosity) may also influence the likelihood and extent of habitat disturbance	Brandt (2000), Grant et al. (2003), Baker et al. (2010), Osterkamp and Hupp (2010), Grant (2012)	Dam height. Sediment transport. Stream habitat complexity and gradient. Tributaries
Hydrologic alteration	Impoundments modify the quantity and timing of stream flows with larger impoundments inducing larger effects. Areas characterized by little prior hydrologic alteration will likely be more sensitive to dam operations. Because dams control streamflow, the potential exists to re-regulate altered stream flows	Moyle and Mount (2007), Poff et al. (2007), Fitzhugh and Vogel (2011)	Degree of regulation. Discharge records. Hydrologic alteration models. Upstream regulation
Water quality alteration	Dams alter temperature, dissolved oxygen, and turbidity levels in addition to intercepting pollutant loads from upstream waterbodies. Existing water quality concerns will determine the potential for exhasterbating or improving water quality conditions with reservoir operations	Hart and Sherman (1996), Webb and Walling (1997), Preece and Jones (2002), Wehmeyer and Wagner (2011), Lessard and Hayes (2003), Wang et al. (2012)	Degree of regulation. Water quality models. Existing water quality concerns, including pollutants
Watershed habitat	Reservoir, road, and powerline construction typically accompanies dam development. Larger reservoirs typically suggest larger effects. Watersheds characteristics (higher relief, soil erosion factor) may predispose terrestrial habitats to increased disturbance from landscape development	Elliot and Hall (1997), Laflen et al. (1997)	Reservoir surface area. Watershed relief and morphology. Soil characteristics, e.g., erosion factor
Watershed condition	Watersheds characterized by intensive disturbance and land-use may pose less threats to biota unless occupied by imperiled species	Esselman et al. (2011, 2013)	Landcover and disturbance. Upstream cummulative impacts
Water use	In areas of intense water usage, reservoir management may increase stress on aquatic organisms by altering the quantity and timing of water delivered downstream through evaporative losses or operations. Upstream regulatory constraints on water timing and availability may govern hydropower operations	L'vovich and White (1990), Gordon et al. (2005)	Water use

Table 1 continued

Environmental concerns	Hypothesis	Sources	Data Needs
Fragmentation			
Habitat connectivity	Dams fragment dendritic river networks; thus, existing connectivity and fragmentation influence population response to additional dam construction. Longer free-flowing reaches, areas of higher connectivity as opposed to shorter fragment lengths, should be avoided for development since they provide corridors for migration and population connectivity; however, different scenarios may exist for diadromous and resident fish	Han et al. (2008), Hoagstrom et al. (2008), Cote et al. (2009), Perkin and Gido (2012)	Dendritic stream network information. Barriers to fish migration
Protected areas			
Conservation lands	Lands protected for their conservation value may be compromised in their ability to support biodiversity, recreation of esthetics following hydropower development. Areas of high conservation value should be avoided	Sowa et al. (2007), Zhao et al. (2010, 2013)	Conservation lands and management regime
Protected rivers	Rivers falling under protection as "free-flowing" also have high conservation value, which may be compromised from dam development	WSRA (1968)	Streams falling under protection
Recreation/esthetics			
Recreation	Hydropower projects have multiple socio-economic effects, including potential changes to recreation, navigable waterways, and surrounding landscapes. While hydropower development provides recreational opportunities, it can also compromise existing recreation	Loomis et al. (1986), Teigland (1999), Hynes and Hanley (2006)	Areas of recreational value
Landmarks	Natural or historic landmarks are important to tourism, esthetics, and human well-being and potentially negatively influenced by dam construction/operation	Teigland (1999), Richardson (2000)	Locations of natural or historic landmarks

regulation or conservation authorities may fail to capture the full potential impacts of development on biodiversity (Pritt and Frimpong 2010). For example, vulnerable fish may also include species that possess sensitive traits, such as migrating long distances to spawn or having specific habitat requirements. In response to large-scale hydropower development in China, Ziv et al. (2012) used a novel approach to model biodiversity loss as a result of reductions in floodplain habitat access for migratory fish. Traits, such as life history characteristics, reproductive strategies, and habitat preferences, may be an efficient way to evaluate landscape-level patterns in fish communities since groups of fish with common characteristics, as opposed to individual species, can be considered collectively (Frimpong and Angermeier 2009).

For species vulnerable to hydropower development, we obtained critical habitats areas (terrestrial and aquatic) for federally listed species under the Endangered Species Act (1973) and spatial distributions of aquatic and terrestrial species of concern (Supplementary Material 1). Species of concern included species that were federally listed or fell under conservation ranking (IUCN 2001) (Supplementary Material 1). We also identified fish species possessing sensitive traits as an indication of potential biodiversity loss. We focused a great deal of attention on fish, rather than other aquatic taxa, because of their well-documented responses to hydropower and influence on regulations regarding hydropower operations. We created a Trait Vulnerability Index (TVI) using six trait combinations, or strategies, deemed representative of fish characteristics that could be vulnerable to hydropower development. The TVI represents the average number or average proportion of species falling into each of the following trait categories: (1) highly migratory, (2) temporally restricted spawning season, (3) habitat specialists, (4) lotic specialists, (5) geographically limited range, and (6) vulnerable life history strategy (Bunn and Arthington 2002; Fausch et al. 2002; Goldstein and Meador 2005; Winemiller 2005; Olden et al. 2006; Moyle and Mount 2007; Mims and Olden 2013) (Supplementary Material 2).

For the national assessment, critical habitats, species of concern, and TVI were summarized for each HUC08 subbasins in the conterminous US (Fig. 2; Supplementary Material 1). However, for the ACF and ACT, we summarized the above metrics within NHD catchments (Fig. 3). Because fish assemblage sampling data not spatially comprehensive (i.e., not all areas are sampled), considerable time was devoted to building species of concern presences and absences using discrete sampling locations and literature (Supplementary Material 2). In addition, species distribution models (SDMs) may be required to predict occurrences at unsampled locations. Rather than develop SDMs for all individual fish species, we constructed trait distribution models to predict the number of species possessing each trait within each catchment (Fig. 3; Supplementary Material 2). Additional information on methods and justification are provided in Supplementary Material 2.

Habitat Alteration Potential

Habitat-related impacts of dams are well documented and typically include alterations to streamflow, impeded sediment transport, altered water quality, and modifications to the terrestrial habitats via inundation (Cushman 1985; Kondolf 1997; Pozo et al. 1997; Bunn and Arthington 2002; Lessard and Hayes 2003; Poff et al. 2007). However, predicting how dams modify habitats at a given location depends on the characteristics of each dam and its operation (e.g., peaking vs run-of-river) (Poff and Hart 2002), and the hydroclimatic and geomorphic context (Grant et al. 2003; McCartney 2009). Several studies have used reservoir storage, acreage, residence time, or a ratio of storage to inflow (degree of regulation—DOR) as surrogates for potential hydrologic and physical impacts to stream habitats (Brown et al. 2009; Chin et al. 2008; Lehner et al. 2011; Kibler and Tullos 2013); however, the extent of impacts may vary depending on the natural or altered state of the river system. Given the difficulty in predicting habitat-related impacts of a particular facility, characterizing the natural topographic and geomorphic context of potential locations for development, along with reservoir characteristics, can provide an indication of the potential for habitat alteration or predisposition to disturbance (e.g., Rojanamon et al. 2009). For example, stream gradient or watershed relief may facilitate or lessen the rate of habitat degradation. Increased erosive potential in higher gradient systems contributes to more rapid coarsening of streambed sediments following reductions in bedload material inputs (Kondolf 1997; Brandt 2000; Gordon et al. 2004). More sinuous channels, however, moderate impacts of dam construction through increased erosive inputs from stream banks or dissipating changes in hydrology through turbulence (Brandt 2000; Gordon et al. 2004). Tributary inputs also ameliorate the downstream impacts of dams by diluting altered hydrology and providing additional sediment inputs (Grant et al. 2003). Higher watershed relief, soil erosive potential, and narrower basins may also facilitate increased terrestrial habitat modification (Elliot and Hall 1997; Laflen et al. 1997).

Landscape planning also requires knowledge of existing infrastructure and anthropogenic disturbances that translate to immediate or cumulative environmental impacts. We could presume that areas with more intense landscape disturbances (as opposed to pristine habitats) may be more appropriate for development unless occupied by vulnerable species, supporting high biodiversity, or supporting

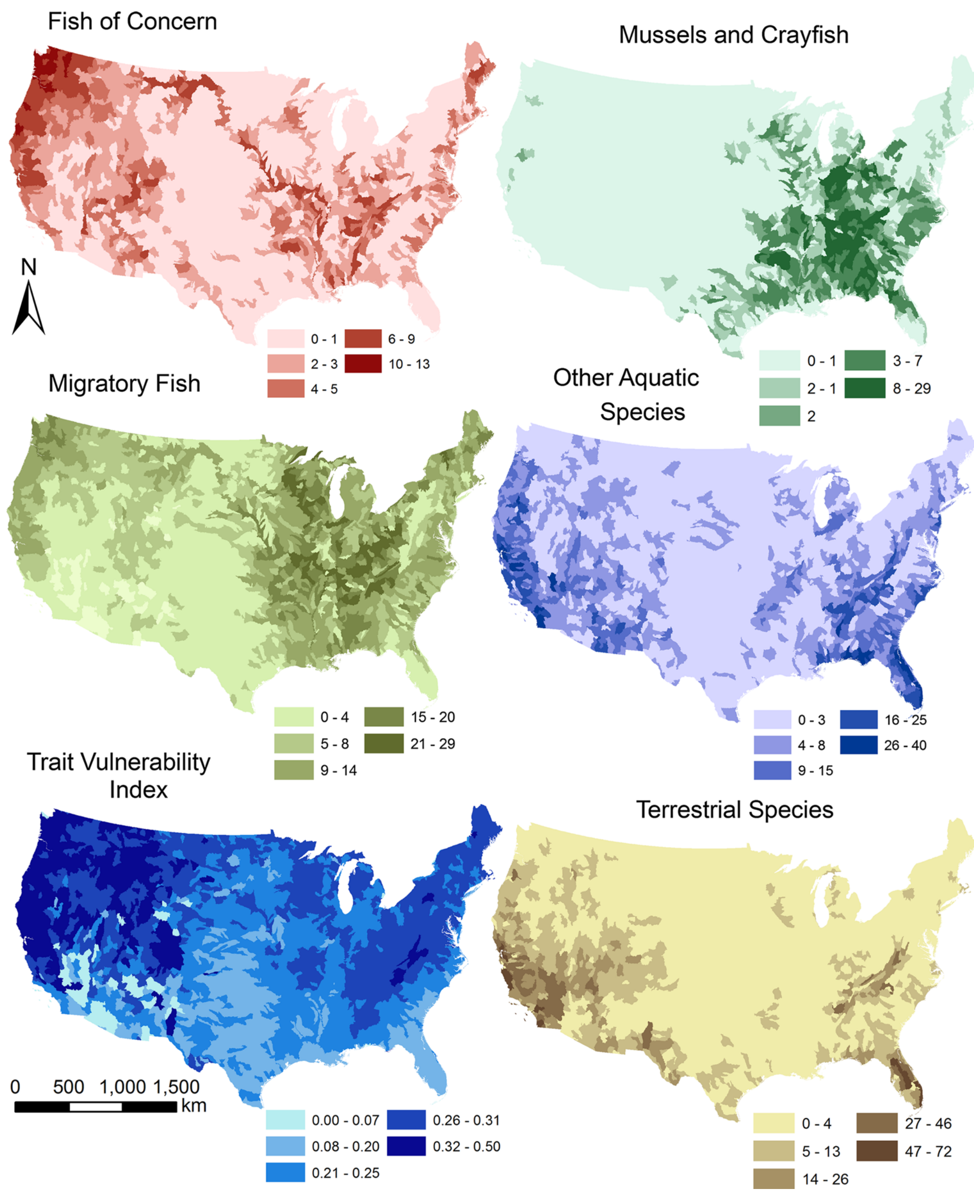


Fig. 2 Biodiversity concerns across the US summarized within HUC08 sub-basins. Biodiversity variables represent species of concern (fish, mussels and crayfish, other aquatic organisms, and terrestrial organisms), number of migratory fish, and the fish Trait Vulnerability Index (as a measure of fish community vulnerability to development), summarized for HUC08 sub-basins

ecosystem services. However, one exception is areas of intense water stress, where additional regulation would compound water demand issues. Water rights for water consumption, appropriation, and availability have become a contentious issue within and across legislative boundaries and basins in the United States. Evaporative losses from reservoirs can be considerable (L'vovich and White 1990) and may modify the timing and amount of water available for downstream users, including stream ecosystems. In addition, upstream regulatory constraints on water timing and availability may govern hydropower operations. Thus, understanding the potential additions to existing water stress associated with each site location is necessary.

We assembled two metrics to assess the propensity for streams and watersheds to disturbance following hydropower development, including: (1) a Stream Channel Disturbance Index (SDI) and a Watershed Disturbance Index (WDI). SDI and WDI were not assessed at the national scale because of the uncertainty in predicting fine-scale habitat changes at the HUC-08 resolution. Within the ACF and ACT, however, natural stream or watershed conditions were assessed and used to develop a Stream Channel Disturbance Index (SDI) and a Watershed Disturbance Index (WDI) for each NHD catchment (Supplementary Material 2). Dam height and reservoir surface acreage have been used as a correlate of degraded habitat conditions downstream of dams and associated terrestrial impacts, respectively (Brown et al. 2009; Tullos 2009); thus, we used each as interaction factors for SDI and WDI indices, respectively (see "Ranking and Prioritization" section).

In order to assess the existing condition or disturbance regime of streams and watersheds, we compiled several

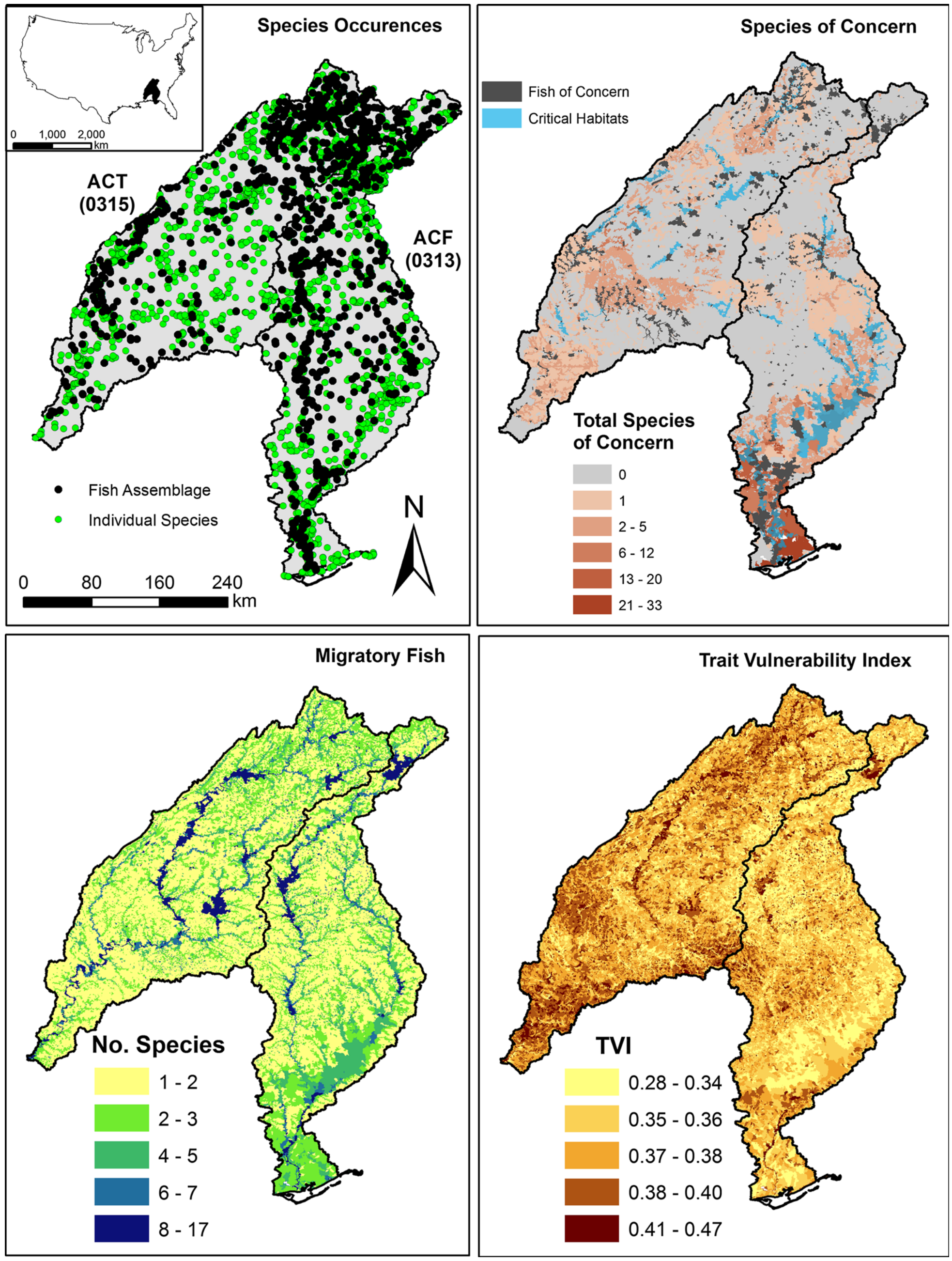


Fig. 3 Biological sampling sites within the Appalachicola–Flint–Chattahoochee (ACF) and Alabama–Coosa–Tallapoosa (ACT) basins (*top left*) are used to summarize the presence of species of concern (*top right*) or predict the number of fish within different trait groups, e.g., migratory fish (*bottom left*). Modeled species richness within multiple trait groups are used to create trait vulnerability indices (*bottom right*)

more variables including: (1) a Hydrologic Alteration Index (HAI), (2) water quality concerns, (3) watershed condition, and (4) water use. Metrics included: (1) Stream channel disturbance spatially predicting hydrologic alteration across stream networks and watersheds has been an under-utilized technique in landscape planning (Richter et al. 1998; Zimmerman et al. 2010; Eng et al. 2013). Approaches to mapping hydrologic alteration includes extrapolating altered hydrologic conditions from stream gages to nearby ungaged streams based on spatial proximity (Richter et al. 1998; Zimmerman et al. 2010) or developing predictive models to extrapolate altered hydrology (Eng et al. 2013). Using several landscape-disturbance variables important in predicting hydrologic alteration (Eng et al. 2013), we derived a HAI for each HUC08 sub-basin in the US (Fig. 4; Supplementary Material 2). Following an approach used by Eng et al. (2013), we predicted hydrologic alteration using models constructed with information on hydrologic alteration and landscape variables at the local scale. A local HAI was calculated for all NHD catchments in the ACT and ACF (Fig. 4; Supplemental Material 2). We used degree of

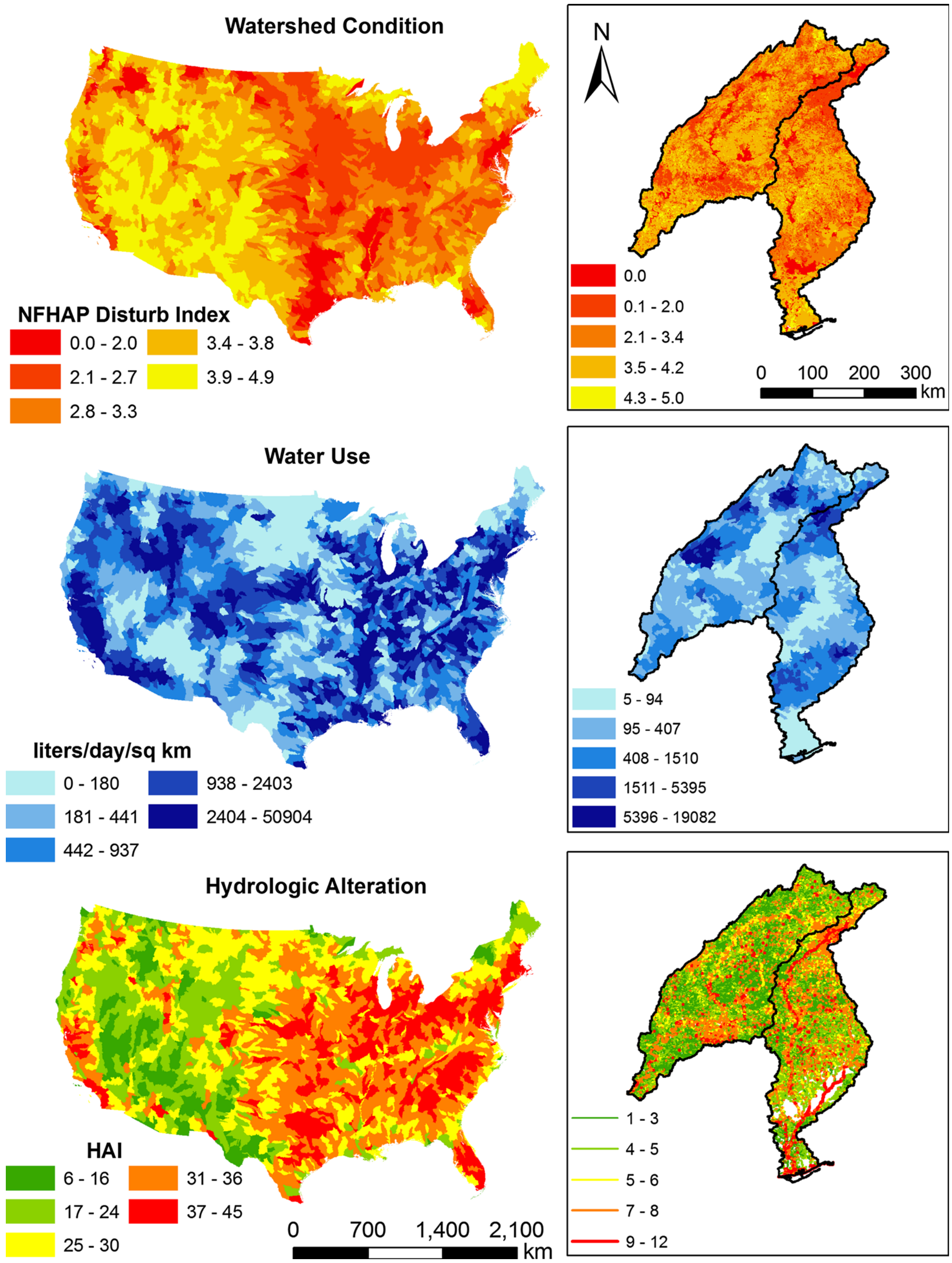


Fig. 4 Pre-existing disturbances in the landscape summarized for the entire US (*left column*) and the ACF and ACT basins (*right column*). Disturbances include watershed condition, as measured by the National Fish Habitat Action Plan (NFHAP) cumulative disturbance Index, water use estimates, and a Hydrologic Alteration Index

regulation (DOR) as the ratio of dam storage to inflow (Lehner et al. 2011) as an interaction factor with HAI (see "Ranking and Prioritization" section).

Other variables were addressed by compiling and summarizing geospatial data. Existing water quality concerns in the landscape were obtained as spatial coverages of impaired waterbodies for the entire United States (EPA 2013; Supplementary material 1). DOR was also used as an interaction factor for the potential to re-regulate poor water quality conditions. Watershed condition was determined using landscape-disturbance information for hydrologic catchments across the United States (Esselman et al. 2011; NFHAP 2013) (Fig. 4; Supplementary Materials 1–2). County-level estimates of water use ($1e^{6}$ gallons day^{-1}) for various consumption categories were obtained from the USGS (Supplementary Material 1) and resummarized as area-weighted averages (l day^{-1} km^{-2}) within HUC08 sub-basins for the US and HUC12 sub-watersheds for the ACF/ACT (Fig. 4) (Supplementary Material 2).

Fragmentation

Within aquatic systems, fragmentation and loss of habitat diversity are the major sources of biodiversity loss

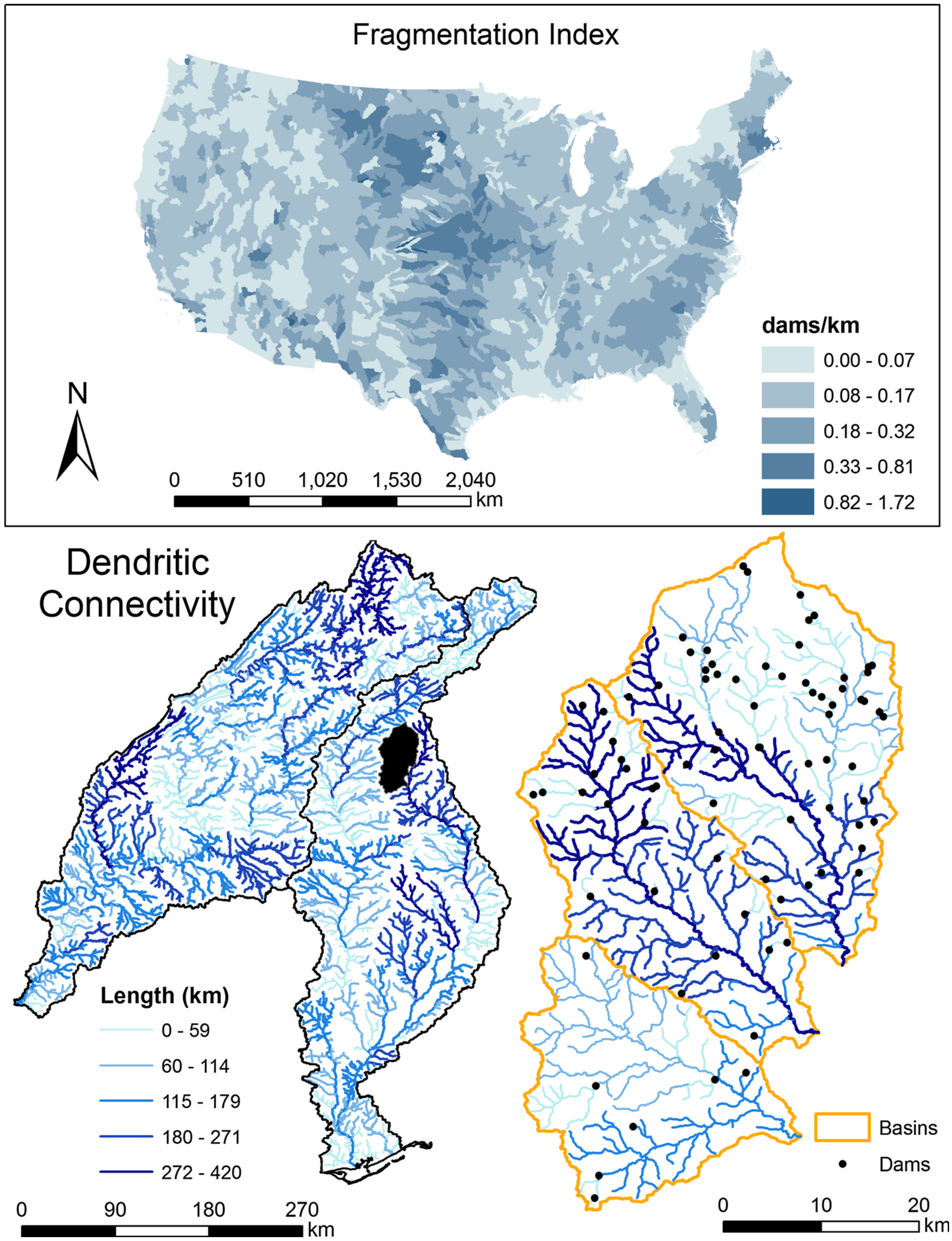


Fig. 5 Examples of different measures of stream fragmentation within the entire US (*top*), within the ACF and ACT basins (*bottom left*), and within three watersheds of the ACF and ACT basins (*bottom right*)

(Vitousek et al. 1997; Jelks et al. 2008). Highly migratory fish species require long free-flowing reaches to complete their life histories (Fausch et al. 2002). However, even fish assemblages predominately composed of non-migratory species can show signs of alteration due to river fragmentation (Perkin and Gido 2012). In addition to fish, the effects of fragmentation are apparent on other aquatic species, such as mussels (Newton et al. 2008).

River fragmentation was addressed in two ways: (1) the number of barriers (i.e., dams) per length of stream in a sub-basin, or (2) a measure of dendritic connectivity (DC) for stream networks for each stream reach (e.g., Cote et al. 2009). For each sub-basin, we calculated a Fragmentation Index (FI) as the total number of dams per length of stream (km) (Fig. 5). Within the ACF and ACT, fragmentation metrics were calculated by discretizing river reaches into segments (i.e., fragments) bounded by upstream or downstream dam locations (Fig. 5). As presented by Cote et al. (2009), DC was calculated as a ratio of fragment length to the total uninterrupted network distance at a given point where a value of 1 indicates a completely free-flowing river system. We also calculated DCE as an indication of estuarine network connectivity where the length of a given discretized river segment is divided by the total downstream river distance to the estuary. A DCE equal to 1 indicates that the river segment runs uninterrupted to the estuary. Although multiple facilities along the mainstem of the ACF and ACT have navigational locks, we did not provide a permeability value for barriers as suggested by Cote et al. (2009) since the ability of many of these structures to provide fish passage has largely been untested (except see Young et al. 2012). In addition, we did not differentiate between potadromous, anadromous, or diadromous scenarios.

Protected Areas

Land alteration is commonly associated with dam construction (Kibler and Tullos 2013; Zhao et al. 2010, 2013). Protected areas are dedicated to preserving biological diversity and natural, recreation, and cultural resources that are managed through legal means (USGS 2014). From an ecological perspective, protected lands represent areas that are refuges, important to supporting biodiversity and ecological integrity. However, from a regulatory perspective, land ownership and designation have widely varying regulations and restrictions on activities, permitting procedures, and development potential. Various levels of protection also regulate development on or near rivers. For example, the Wild and Scenic River Act in the US, was enacted to protect certain rivers that possess remarkable ecological and esthetic values as free-flowing (WSRA 1968). In addition, the Nationwide Rivers Inventory lists free-flowing river segments that are considered to possess one or more “outstandingly remarkable” natural or cultural values (NPS 2011).

Spatial coverages of conservation lands and their associated conservation status, US Wild and Scenic River Systems, and US National River Inventory were obtained from multiple sources (Supplementary Material 1) and summarized differently for the US and local scales (Fig. 6). The conservation status (GAP status) ranges from 1 to 4 (Fig. 7) and indicates the degree of protection toward a natural state and allowable resource use with 1 representing areas managed strictly for biodiversity purposes (e.g., national park) and 4 representing areas supporting multiple extractive uses (e.g., military lands). Presumably, biodiversity and species of concern would correlate with GAP status; however, densities of federally listed and imperiled species are three times higher on military lands than other federal lands (Stein et al. 2008). Hence, development on any conservations lands, regardless of GAP status, could have equally negative effects.

Recreation and Esthetics

Identifying areas and waterways of the landscape that are highly valued by the public is an essential aspect of planning for development. While recreation and esthetic factors might seem misplaced in our analysis, there is considerable overlap with ecological and biophysical effects of hydropower development. For example, the economic value of recreation (e.g., eco-tourism) has been shown to correlate with areas of high biodiversity (Gössling 1999). Likewise, recreational boating, hunting, fishing, and hiking typically rely on areas of the landscape that are protected for conservation purposes (Cordell et al. 2013). Quantifying the esthetic value from the landscape is complex (Gobster et al. 2007), yet identifying areas established as recreational or natural landmarks provides a straightforward approach to determining potential concerns.

Because the PAD-US database includes recreational opportunities, such as hiking, camping, and historical landmarks, we did not explicitly address these issues separately from protected lands. For the entire US, point locations of fishing access areas, boat ramps, and waterfalls were compiled to represent types of aquatic and terrestrial recreation (Fig. 7; Supplementary Material 1). Launch and take-out point locations for whitewater boating runs across the US were obtained from the National Whitewater Inventory (Supplementary Material 1) and used to create whitewater boating routes (Fig. 8; Supplementary Material 2).

Step 4: Spatial Footprint

At the national scale analysis, we used HUC08 sub-basins as our spatial unit to summarize environmental datasets (Table 2). Many datasets were summarized as counts (e.g., species of concern) whereas for others, such as critical habitats or protected lands, we summarized polygons and polylines as total length (km) or area (km^2) represented within each HUC08 sub-basin (Table 2). Other environmental data, such as fishing access locations, were summed as counts (Table 2).

In contrast to the national analysis, local-level analyses require determining environmental concerns for a particular hydropower project. For the ACF and ACT, we developed a hydropower spatial footprint composed of three elements: a dam, an inundated area, and a downstream section (i.e., tailwater) (Fig. 8). Given that dams varied little in height, we postulated that minimum tailwaters length should be 15 km to capture most immediate effects (Ellis and Jones 2013). However, because tributary inputs ameliorate the downstream effects of an impoundment (Ward and Stanford 1983; Ellis and Jones 2013), varying tailwater lengths on the basis of tributary junctions would be a reasonable attempt to approximate downstream concerns. Because NHD flowlines were discretized on the basis of tributary junctions for a given basin (HSC 2013),

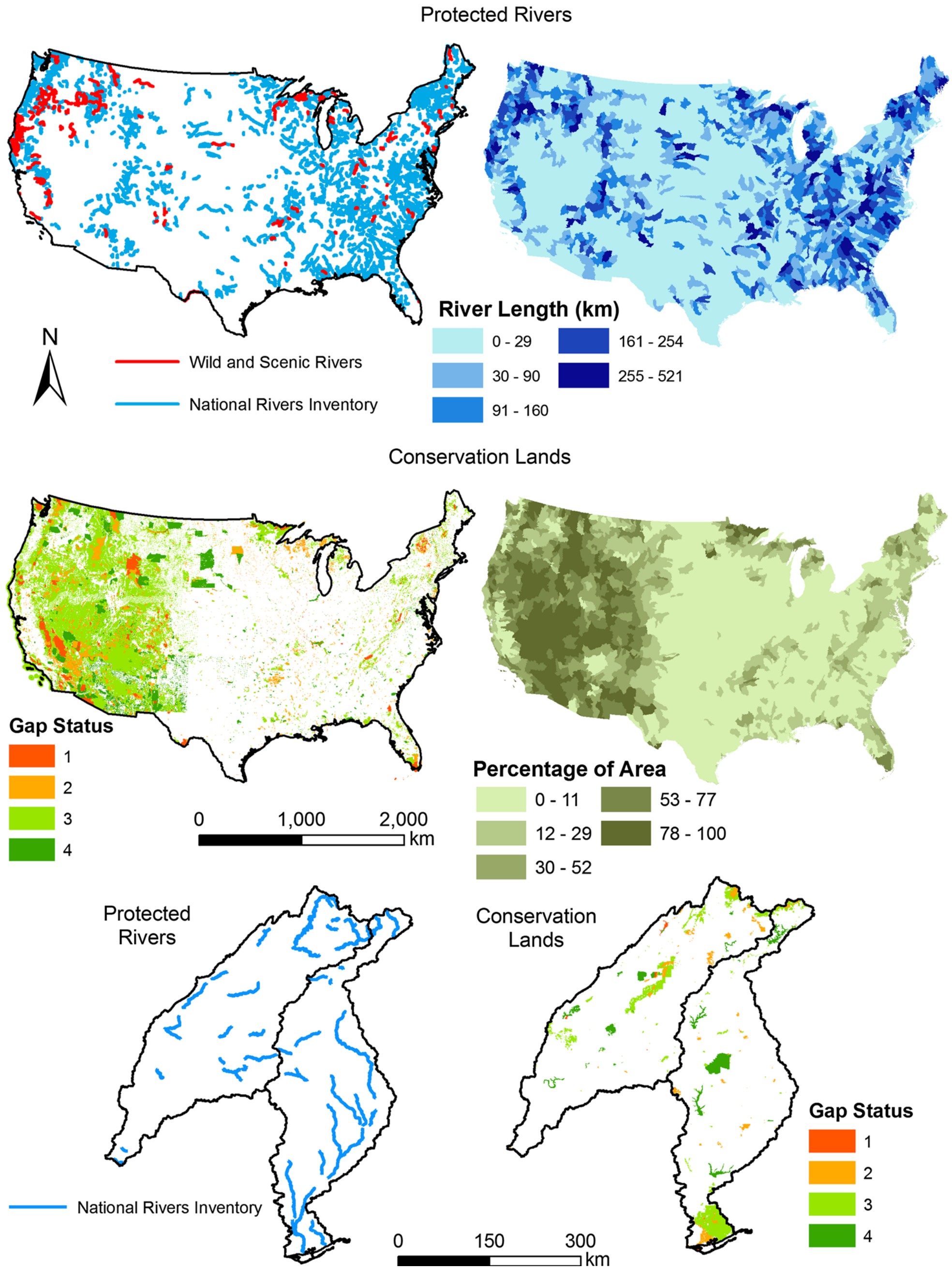


Fig. 6 Spatial distribution of protected rivers and protected lands (conservation lands) in the entire US (*top* 4 maps) and within the ACF and ACT basins (*bottom* 2 maps). For the entire US, the length of protected rivers and acreage of protected lands were summarized within HUC-08 sub-basins

and thus, the length of NHD flowlines vary considerably. Thus, tailwater lengths were determined by iteratively selecting NHD flowlines downstream of a potential dam until the cumulative length reached 15 km. In the ACF and ACT, tailwater lengths ranged from 15 to 44 km.

The spatial extent (i.e., buffer size) at which environmental concerns may be realized at a site depends upon the unique combination between each element (e.g., dam, reservoir, tailwater) and the environmental variable under consideration (Table 3). We established a maximum buffer size for each unique element-environmental data combination based on the literature review of buffer sizes for approximating environmental effects (e.g., Bohlen and Lewis 2009; Rojanamon et al. 2009; Soussan et al. 2009; Kibler and Tullos 2013; Zhao et al. 2010, 2013) (Table 3). Within the maximum allowed buffer size, we varied buffers across the three elements to determine spatial uncertainty in environmental attribution (Fig. 8). Larger buffers were associated with dams due to uncertainties in potential road or power line development. Each maximum buffer was divided into 5 equidistant-incremental buffer sizes (i.e., concentric rings), and environmental data were iteratively attributed to each increment. For example, a 4,000-m maximum buffer size would include 0-, 1,000-, 2,000-,

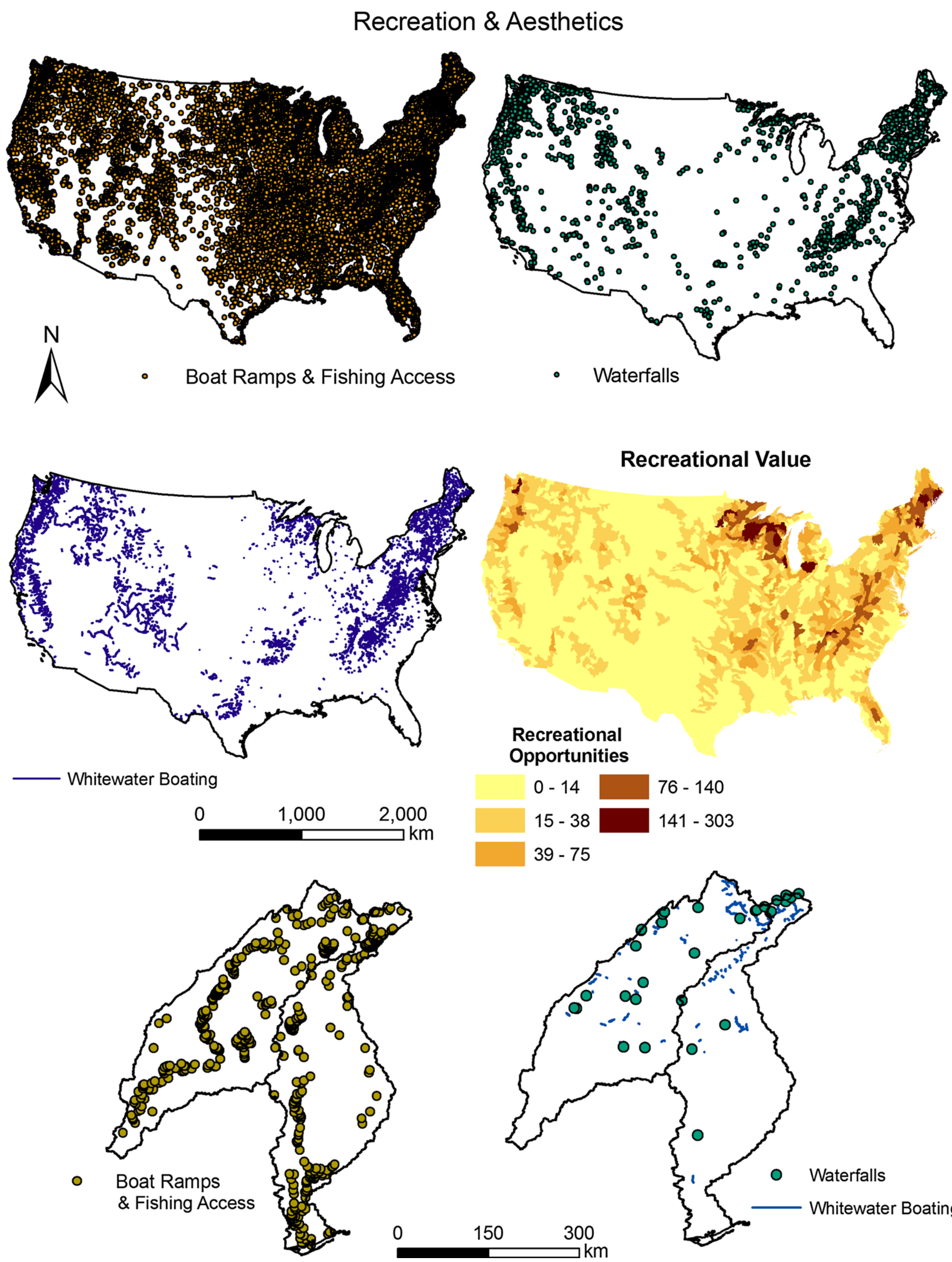


Fig. 7 Locations of recreational opportunities in the entire US (*top* 4 maps) and within the ACF and ACT basins (*bottom* 2 maps). For the entire US, the number of recreational opportunities were summarized within HUC-08 sub-basins

3,000-, and 4,000-m incremental buffers (Table 3). The method used to attribute (e.g., intersection, spatial join) and summarize environmental data to each site depended on the environmental data layer (Table 3).

Step 5: Ranking and Prioritization

We provide different ranking schemes for the US and the local scale. For the US, we conduct a reserve-design analysis, where reserves are preserved from development. For the ACF and ACT basins, we conduct a multi-metric scoring technique to show how cumulative indices may represent or misrepresent important individual environmental components. For each ranking scheme, we provide alternative scenarios to reflect varying expertise and perspectives and to highlight the need for stakeholder involvement. Our scenarios reflect our assumptions only and are not meant to represent any organizational or agency standpoint or perspective on future hydropower development. They are provided as examples only.

National Scale

Across the US, we used basins as planning units in developing reserve-design scenarios using Marxan version 2.43 software (Ball et al. 2009). The objective function of

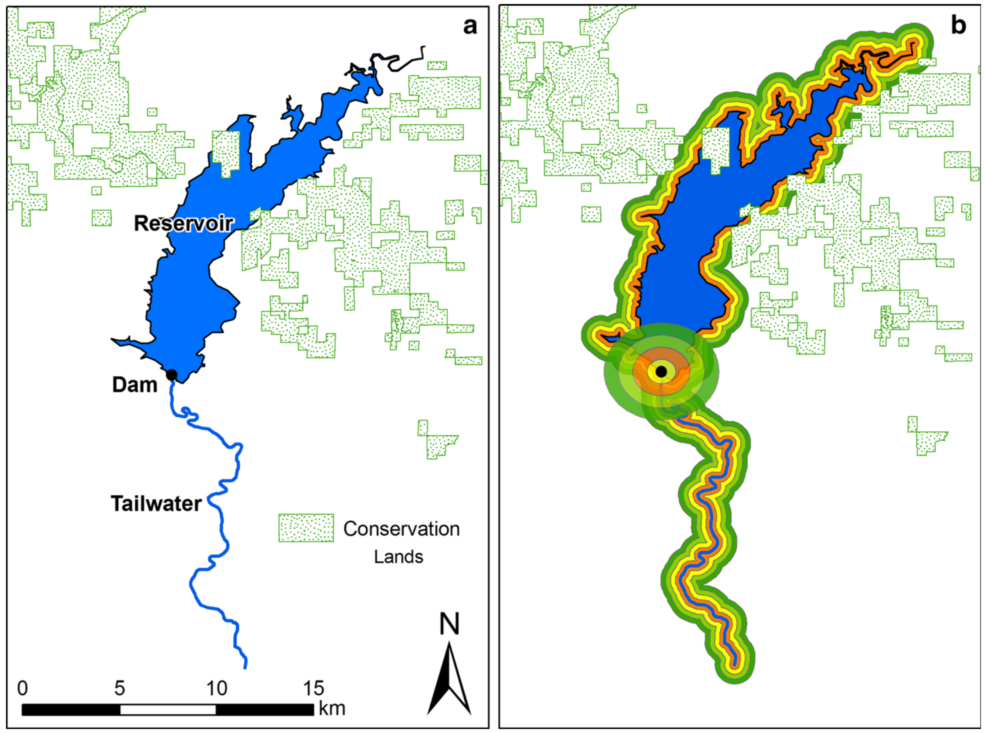


Fig. 8 **a** Example of the spatial footprint of a potential hydropower project, depicted by a dam, reservoir, and tailwater. **b** Uncertainty in interactions between the site and environmental spatial data (e.g., conservation lands) can be captured using multiple concentric ring buffers around the spatial footprint

Marxan is to ensure a minimum set of biodiversity characteristics are represented in reserves at the lowest minimum cost (Game and Grantham 2008; Ball et al. 2009). In this case, reserves are identified as areas where hydropower development should be avoided relative to potential energy gained as opposed to identifying areas for permanent protection relative to minimum reserve area (McDonnell et al. 2002). While this may seem to be a minute difference, our approach is fundamentally different than standard approaches to running Marxan. First, because we are not attempting to achieve traditional reserve designs, as an important consideration is that inclusion of a given planning unit into a reserve should be inclusive rather than exclusive, given that the overall intent is preventative. Thus, the location, frequency, and characteristics of planning units designated as "pre-existing" reserves is extremely important to the final solution. Second, as opposed to being dependent upon area, cost is a function of (1) potential energy compromised by including a particular planning unit in a reserve and (2) the level of existing anthropogenic disturbances within each planning unit. Last, within hydrologic connectivity among planning units is more important than spatial proximity. Thus, when minimizing reserve areas, planning units in close proximity should only be prioritized in the reserve design unless occurring within similar watersheds.

Conservation features within each HUC-8 sub-basin included all species of concern, trait vulnerability indices, protected land and river coverage, and recreation areas. Cost for each planning unit was either a function of potential MW capacity, habitat disturbance (with emphasis on fragmentation), or both characteristics (Supplementary Material 3). Connectivity (i.e., analogous to boundary length) among planning units was assessed as sub-basins that shared similar basins or sub-regions (Supplementary Material 3). Sub-basins sharing a boundary and occurring within the same basin or sub-region were ranked higher than basins not sharing any boundary. Sub-basins not occurring within similar sub-regions were considered not connected and excluded from the boundary file. The initial status of reserves was determined using protected land coverage and levels of representation among different conservation features and ranged from very inclusive, to very exclusive, to no initial reserves (Supplementary Material 3). Different reserve designs were simulated in Marxan using 48 different scenarios by varying cost, the relative importance of connectivity, and the initial reserve design. Each scenario included 10 iterations each. Detailed methods on data preparation and running Marxan are provided in Supplementary Material 3.

Higher MW capacity was pervasive in the eastern US but constrained to major river systems within arid regions of the western US (Fig. 9). The outcome of different reserve designs was highly dependent upon each scenario and we highlight 2 of the 48 as examples (Fig. 9). In scenario A, optimal reserves were simulated based on the following criteria: (1) a semi-exclusive-initially-strict set of pre-defined reserves (i.e., planning unit status), (2) strong

Table 2 Approach to summarizing environmental data at the national scale (witihin HUC08 sub-basins)

Category	Component dataset	Raw data type	Summarization
Biodiversity	Critical habitats	Line, polygon	Length (km), Area (km^2), Count
	Species of concern	HUC08	Count
	Vulnerable fish traits	HUC08	TVI
Habitat alteration	Hydrologic alteration	HUC08	HAI
	Water quality (303d list)	Point, line, polygon	Length (km), area (km)
	NFHAP catchment disturbance	HUC08	NFHAP Index
	Water use	Counties	l day^{-1} km^{-2}
Fragmentation	Fragmentation Index	Point, line	Index (dams km^{-1})
Protected areas	Conservation lands	Polygon	Area (km^2)
	Wild & Scenic Rivers	Line	Length (km)
	National Rivers Inventory	Line	Length (km)
Recreation	Fishing access/boat ramp	Points	Count
	American Whitewater Boating	Lines	Length (km)
	Waterfalls	Points	Count

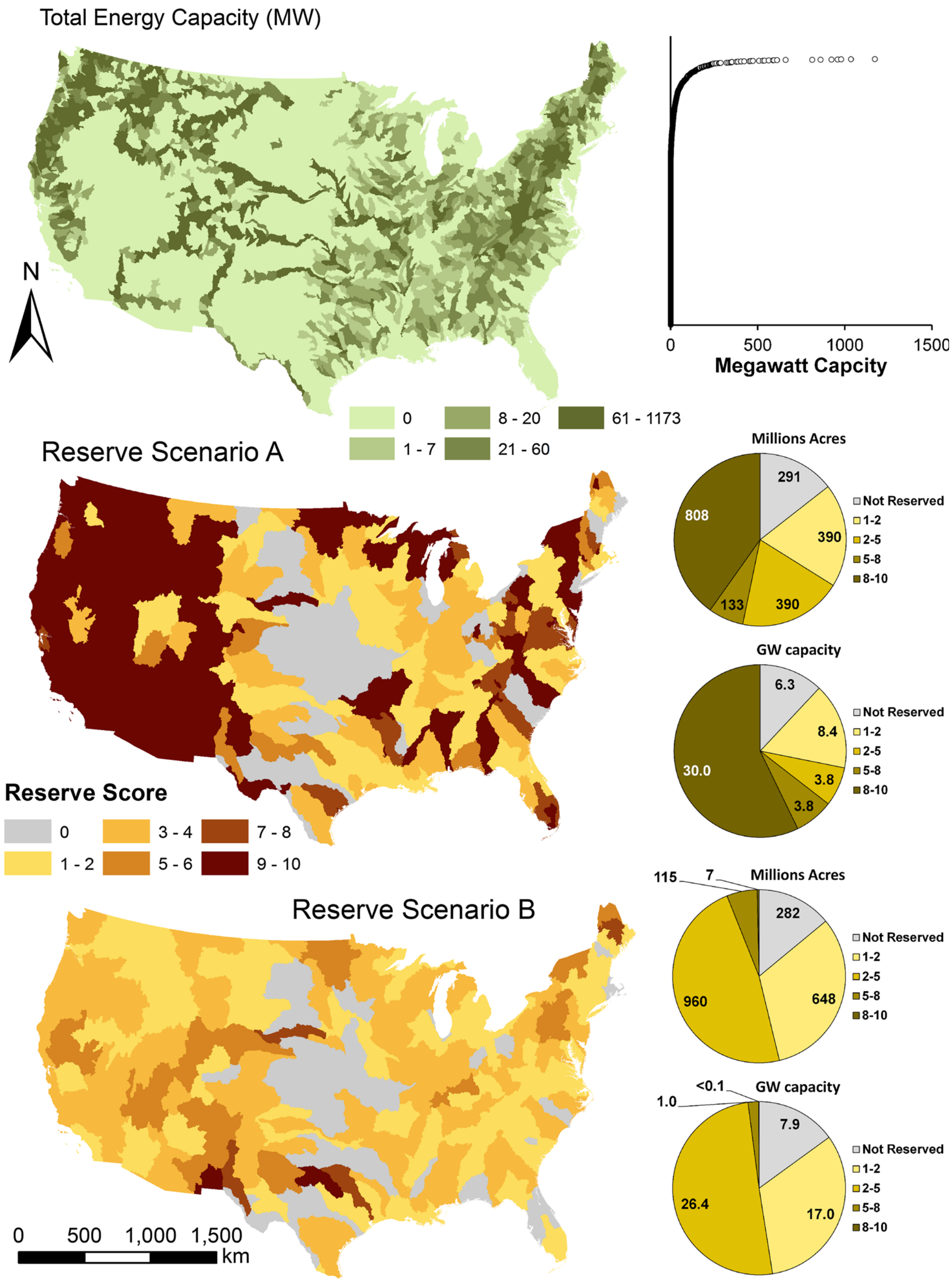
TVI Trait Vulnerability Index, *HAI* Hydrologic Alteration Index, *NFHAP Index* National Fish Habitat Action Plan Index

Table 3 Approach to summarizing environmental data at the local scale (site-level)

Category	Component dataset (V)	Buffer width (m)			Local reach	Summarization
		Dam	Reservoir	Tailwater		
Biodiversity	Critical habitats	4,000	1,000	1,000	–	Count
	Species of concern	–	–	–	NHD	Count
	Vulnerable fish traits	–	–	–	NHD	TVI
Habitat Alteration	Stream channel disturbance	–	–	–	NHD	SDI
	Hydrologic Alteration Index	–	–	–	NHD	HAI
	Water quality (303d list)	500	500	500	–	Count
	Watershed habitat disturbance	–	–	–	NHD	WDI
	NFHAP disturbance	–	–	–	NHD	NFHAP Index
	No. of upstream dams	–	–	–	NHD	Count
	Population density	–	–	–	NHD	Individuals km^{-2}
	Water use	–	–	–	HUC12	l day^{-1} km^{-2}
Fragmentation	Fragment length	–	–	–	NHD	Length (km)
	Dendritic connectivity					DC Index
	Estuarine dendritic connectivity	–	–	–	NHD	DCE Index
Protected areas	Protected lands—PAD-US	2,500	1,000	1,000	–	Area (km^2)
	Wild & Scenic Rivers	2,500	1,000	1,000	–	Count
	National Rivers Inventory	2,500	1,000	1,000	–	Count
Recreation	Fishing access/boat ramp	500	500	500	–	Count
	American Whitewater Boating	500	500	500	–	Count
	Waterfalls	2,500	1,000	1,000	–	Count

TVI Trait Vulnerability Index, *SDI* Stream Channel Disturbance Index, *HAI* Hydrologic Alteration Index, *WDI* Watershed Disturbance Index, *NFHAP Index* National Fish Habitat Action Plan Index, *DC Index* Dendritic Connectivity Index, *DCE Index* Estuarine Dendritic Connectivity Index

Fig. 9 Spatial distribution of energy capacity (*top left*) and associated MW distribution (values ranked from low to high) (*top right*) in the US. Alternative reserve design solutions are represented by reserve scores, i.e., number of times a unit was selected out of ten iterations (*middle and bottom left*). The total area within different reserve scores and associated MW capacity are also provided (*middle and bottom right*)



emphasis on topological connections among basins (BLM = 0.75), and (3) cost estimated by combining potential energy with disturbance estimates. For scenario B, optimal reserves were simulated similar to A, except that pre-defined reserves were not strict (i.e., pre-assigned reserves may or may not be included in the final reserve design). A considerable number of planning units were selected for reserves at least 90 % of the time in scenario A, whereas very few units were selected >50 % of the time in scenario B (Fig. 9). While the total reserve area was far greater in scenario A than B (in terms of units repeatedly selected), the total area and GW capacity of units never selected as reserves was similar. Differences between scenario A and B illustrate the importance of defining the "protection" status of planning units prior to running Marxan. Furthermore, these results emphasize the importance of stakeholder involvement in running scenarios.

Local Scale

The ACF and ACT basins provided an example to explore environmental concerns within the spatial footprint of sites.

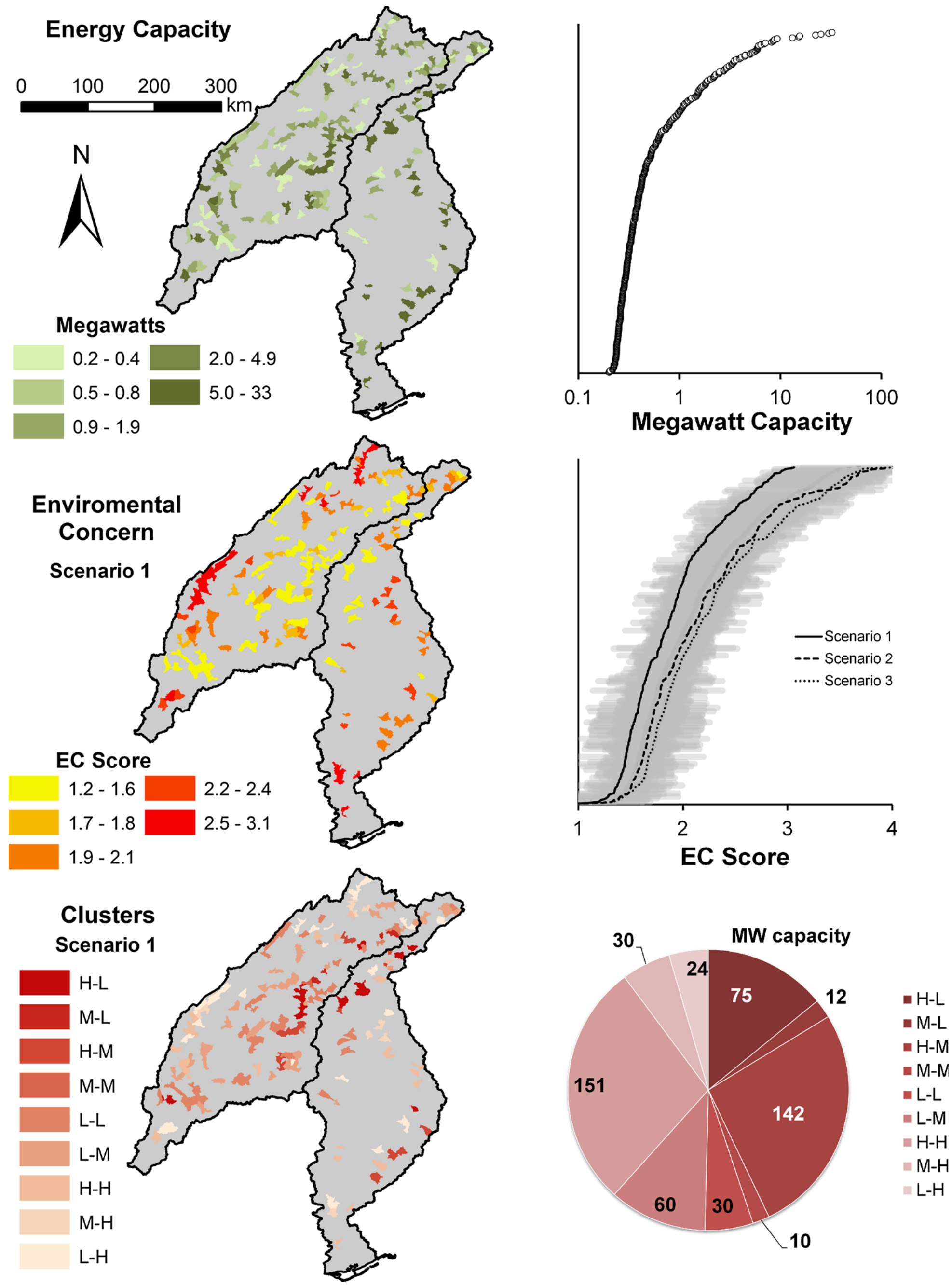


Fig. 10 Spatial distribution of energy capacity (*top left*), environmental concern (EC) scores (*middle left*), and clusters combining energy capacity and EC scores (*bottom left*) are mapped according to HUC12 catchments in the ACF and ACT basins. Numeric distributions of energy capacity (*top right*) and EC scores (*middle right*) represent values ranked from lowest to highest. EC scores represent solutions for three alternative scenarios (see Step 5, Local Scale). *Gray shaded areas* represent standard deviations for all three scenarios based on variation in buffers, the *darker the area* the more overlap among scenarios. EC scores and clusters were mapped for scenario 1. The *pie chart* represents MW capacity associated with each cluster

As opposed to using reserve-designs, we explored developing cumulative Environmental Concern (EC) scores as composite values that summarized all environmental categories, but with the added flexibility of visualizing individual score components in diagrams. EC scores were determined using the following equation:

$$\mathrm{EC}_{iej} = a * [\mathrm{BD}]_{iej} + b * [\mathrm{HA}]_{iej} + c * [\mathrm{FR}]_{iej} + d * [\mathrm{PA}]_{iej} + f * [\mathrm{RA}]_{iej}, \quad (1)$$

where (EC_{iej}) is the environmental concern score for the *i*th site, *e*th element (e.g., dam, tailwater, reservoir), and *j*th buffer size based on scaled estimates of each category. Individual categories include biodiversity (BD), habitat alteration potential (HA), fragmentation (FR), protected areas (PA), and recreation/esthetics (RA). Coefficients *a*, *b*, *c*, *d*, or *f* can be used to assign weights to each category if desired. Each category can be broken into separate equations. To provide an example of one component, the

Table 4 Results of three scenarios of environmental concern scores calculated for the local scale (site-level)

Cluster	Scenario 1		Scenario 2		Scenario 3	
	Sites	MW	Sites	MW	Sites	MW
High–high	16	151	15	143	18	143
High–low	12	75	10	64	12	50
High–moderate	27	142	30	161	25	175
Low–high	62	24	61	23	59	23
Low–low	77	30	78	30	80	28
Low–moderate	159	60	159	61	159	63
Moderate–high	20	30	22	33	21	32
Moderate–low	9	12	10	13	6	7
Moderate–moderate	8	10	5	7	10	13

Clusters represent energy and environmental concern (EC) score combinations. EC scores were calculated using the following scenarios: (1) all categories weighted the same, (2) Biodiversity (BD) category weighted as 2 and other categories as 1, (3) Fragmentation (FR) weighted as 2 and other categories as 1

scoring process for the BD category is demonstrated as follows:

$$\mathrm{BD}_{iej} = \sum_{h=1}^{7} [V]_{iejh}, \quad (2)$$

where BD_{iej} is the sum of scores from seven component datasets ([V]), which, in the case of biodiversity, included critical habitats, five species of concern variables (fish, mussels and crayfish, amphibians, other aquatic species, and terrestrial species), and the fish Trait Vulnerability Index. Thus, [V] is the value from the hth component dataset for each site, element, and buffer size.

The outcome of V depends on the rules that determine how the directionality in values for environmental data translated into negative or positive environmental effects for hydropower development (Supplementary Material 4). The rule-based ranking procedure incorporates uncertainty in the spatial representation of environmental concerns based on variation in spatial proximity. In addition, V depends on interactions with dam and reservoir characteristics, such as dam height and residence time, respectively. For example, improving water quality conditions depends on the extent of water quality concerns and the potential for a dam to manipulate water quality. A detailed example of the scoring process and rules are provided in Supplementary Material 4.

Based on variation in spatial proximity, we calculated an average $\overline{\mathrm{EC}}_i$ score and variation, σ_i, for each of the ith sites. Hence, variation in spatial representation can be explicitly reported. In addition, users may select minimum or maximum values for sites based on the occurrence of sensitive environmental features. We incorporated energy values and EC scores into composite values using clustering. All sites were ranked according to energy and EC scores. According to Department of Energy, new development is classified according to a 1 MW threshold, where <1 MW capacity is "Low" (Kao et al. 2014) and >1 MW as "High." However, we further classified sites with 1–2 MW as "Moderate" and sites >2 MW as "High," as these were above the 90th percentile values for small hydropower in the ACF and ACT basins.

We developed three scenarios for assessing EC scores in the ACF and ACT using Eq. 1. For scenario 1, EC scores were developed with all categories weighted equally. Scenario 2 calculated EC scores with BD having a weight of 2 whereas all other categories received 1. Scenario 3 weighted FR as 2 compared to 1 for other categories. Following calculation of EC for each scenario, sites were classified as "Low," "Moderate," or "High," by their environmental concern values using the following categorization: <25 %'tile = "Low"; 25 %'tile-75 %'tile = "Moderate," >75 %'tile = "High." Energy and environmental data were combined to create a 9-tier classification of energy–environmental concern values (e.g., high–low, low–high). Because of the sensitivity of showing exact locations of potential sites, we mapped clusters within HUC12 sub-watersheds. To evaluate the contribution of individual components to each EC score, spider diagrams were also produced for each site.

Within the ACF and ACT basins, the majority of high energy areas confined to larger river systems (Fig. 10). Distributions of EC scores for all scenarios followed a similar pattern (Fig. 10). We mapped the solution for scenario 1 (Fig. 10). Higher EC scores were present in the headwaters and lower portions of both basins, the eastern edge of the ACF, and the western edge of the ACT (Fig. 10). Interestingly, the highest density of sites falling within the "High–Low" cluster was centralized to the middle of each basin whereas "Low–High" sites were scattered throughout the basins. In all scenarios, the largest proportion of energy was found in the "High–High" and "High–Moderate" categories. Thirty-four sites changed clusters from scenario 1 to scenario 2 whereas 52 sites changed clusters from scenario 1 to scenario 3 (Table 4). Total MW capacity in the "High–Low" category progressively degreased from scenario 1 to scenario 3 (Table 4), suggesting that biodiversity and fragmentation issues may be more prevalent in areas with higher energy density. Although clusters are useful in evaluating overall trends, coarse clustering may overshadow differences in individual components that may be significant when evaluating potential sites. Spider diagrams can be used to compare individual components that make up cumulative EC scores. In order to show key differences in types of environmental concerns that may not be apparent in coarse

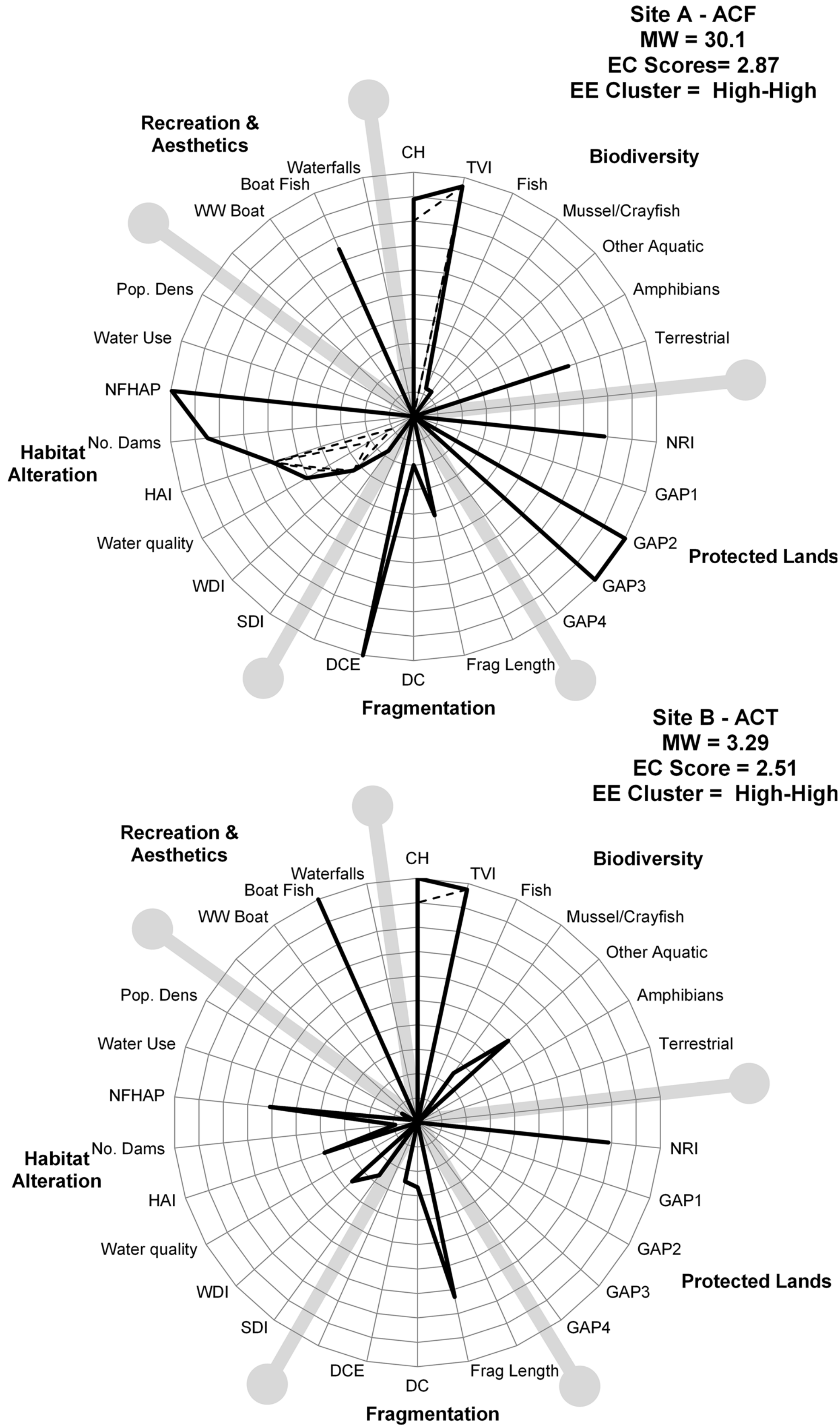


Fig. 11 *Spider diagrams* depicting values for components of each of the five environmental categories. Two potential hydropower sites as examples for comparison. *Dark lines* represent maximum values whereas *dashed-lines* represent values for smaller buffers

clustering, we selected two sites that had similar EC scores and both fell within the high–high cluster (Fig. 11). Site A occurs in the lower ACF basin whereas site B occurs in middle portion of the ACT basin. On the basis of energy alone, site A would be the superior choice for development relative to site B. However, evaluating the individual components reveals some significant differences. Site A has higher values for protected lands, but higher disturbances (Fig. 11). In addition, although total fragment length is higher for Site B, site A has a DCE = 1, which indicates a free-flowing connection to the ocean (Fig. 11). Hence, social roundtable discussion is required to refine relative importance of various environmental concerns.

Social Roundtable

An obvious conclusion from the assessment above is that alternative scenarios are assumption-driven and reveal very different and conflicting results. However, multiple scenarios are needed to represent varying expertise, opinion, and stakeholder interest. While the importance of stakeholder involvement is evident during ranking and prioritization, stakeholder groups must be present during in the entire process, including hypothesis generation, data compilation, and summarization since these steps will influence final results. As one example, not all sources of data are common knowledge and may be overlooked in any analyses. Additionally, the method in which metrics are summarized is paramount in final solutions. Even simple procedures, such as classifying energy capacity into categories, will influence the interpretation of the analysis. While 1 MW was an important threshold in our analysis, it should be noted that globally, 50 MW has been used as a threshold to define small versus large hydropower (Kibler and Tullos 2013).

The interpretation of results will vary according to the audience and the scale of the analysis. For example, at the national scale, reserve areas can be identified as a coarse representation of higher environmental risk and higher environmental mitigation costs associated with energy development. These assessments may influence policy decisions and investments for future energy development scenarios. The national-level analysis also provides a schematic to prioritize basins for future research (Fig. 10), such as the potential to optimize increasing energy and environmental mitigation. For the local-scale assessment, areas of high energy–low environmental concerns can be prioritized as locations of finer-scale GIS analyses (Fig. 11). In addition, the local-scale assessment provides a coarse precursor to future environmental impact assessments. However, both scales share a similar product in that they foster stakeholder discussions regarding hydropower development above the level of site-by-site assessments.

Conclusions and Limitations

Many countries are facing the need to meet growing energy demands while also reducing CO_2 emissions, thus making hydropower development a preferred alternative to fuel-based sources of energy (Grumbine and Pandit 2013). In order to increase national energy budgets, the current pattern of hydropower development in many countries is widespread as opposed to occurring only in individual locations (e.g., 292 dams planned for Himalayan region, India) (Grumbine and Pandit 2013). Within countries undergoing intense development, large numbers of planned hydropower facilities pose new challenges for EIAs because assessment needs extend well beyond the scope of traditional site-by-site approaches (Tullos 2009; Elerwein 2013).

While we attempt to provide a comprehensive evaluation of ecological and biophysical effects induced by hydropower development as an example, many socio-economic and geopolitical factors should also be considered in analyses before results are used within social roundtable discussions. Our analysis was based on comparing the effects of developing alternative facilities; thus, another limitation of our approach is that we did not quantify cumulative impacts of multiple developments, such as additive impacts of developing more than one facility. However, the approach presented herein provides a starting point for developing sophisticated and comprehensive approaches to inform widespread hydropower development. There are other foreseeable obstacles to applying our approach in many countries, mostly related to data availability, differences in regulatory contexts, and variable emphasis on specific environmental issues. While the US provides a convenient opportunity to apply the spatial approach, it should be noted that intensive data drives (including literature), geoprocessing, and statistical modeling were required to adequately represent many concerns. Because data availability will influence what and how environmental factors are assessed (e.g., Nu River Basin, Kibler and Tullos 2013), we urge researchers and conservation planners to consider creative approaches to representing environmental concerns in the landscape. In addition, the concerns we present may not be representative of concerns other countries are facing with large hydropower expansion, such as human displacement (Soussan et al. 2009) and compromised food security related to loss of floodplain habitat (Ziv et al. 2012).

Last, any procedure that ranks or prioritizes development requires making assumptions. Great care should be used in interpreting our results or in using our results to insinuate lesser or greater environmental effects to different locations. Our assumptions reflect one of many possible scenarios; thus, future research is needed to incorporate additional uncertainty in environmental concerns on the basis of varying values and perceptions. Our hope is that the methodology presented can be useful to future applications and will foster discussion of how to ensure environmentally sustainable energy development. Despite common belief, there are opportunities for mutual benefits between hydropower development and environmental protection, as long as stakeholders are involved in the discussion. One such example is the Penobscot River restoration project (Maine, USA), where improved passage and removal of dams in the lower portion of the basin freed up almost 1,000 miles of habitat for anadromous species, such as Atlantic salmon and shortnose sturgeon (PRRT 2014). Losses in energy in the lower portion of the basin were offset by increasing energy production at six upstream dams, leading to a total net increase in energy production (PRRT 2014). Likewise, powering existing non-powered dams and building energy efficient dams with higher energy output in environmentally degraded areas could be balanced by restoration of aquatic habitats, conservation land acquisition, or removal of dams elsewhere. Environmental scientists and stakeholders should be engaged in conversations over future development as to guide future research efforts toward analyses aimed at finding solutions to optimize energy and environmental sustainability.

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