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Advancing Environmental Flow Science: Developing Frameworks for Altered Landscapes and Integrating Efforts Across Disciplines

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Abstract Environmental flows represent a legal mechanism to balance existing and future water uses and sustain non-use values. Here, we identify current challenges, provide examples where they are important, and suggest research advances that would benefit environmental flow science. Specifically, environmental flow science would benefit by (1) developing approaches to address streamflow needs in highly modified landscapes where historic flows do not provide reasonable comparisons, (2) integrating water quality needs where interactions are apparent with quantity but not necessarily the proximate factor of the ecological degradation, especially as frequency and magnitudes of inflows to bays and estuaries, (3) providing a better understanding of the ecological needs of native species to offset the often unintended consequences of benefiting non-native species or their impact on flows, (4) improving our understanding of the non-use economic value to balance consumptive economic values, and (5) increasing our understanding of the stakeholder socioeconomic spatial distribution of attitudes and perceptions across the landscape. Environmental flow science is still an emerging interdisciplinary field and by integrating socioeconomic disciplines and developing new frameworks to accommodate our altered landscapes, we should help advance environmental flow science and likely increase successful implementation of flow standards.

Keywords Environmental flows · Human influence · Altered landscapes · Economic value of water

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Introduction

Societies exploit the benefits of freshwater and place extensive stress on aquatic ecosystems. River modification was documented as early as 5000 years ago via civilization of ancient Egypt and Mesopotamia (Wootton 1990). Since that time, aquatic ecosystems have been channelized, dammed, dredged, leveed, and pumped to reduce flooding, promote river commerce, exploit hydropower and meet industrial, agricultural, and municipal needs. Although human needs are a necessary component of water use, the extensive pressures on our global aquatic systems have become clear with approximately 20 % of all freshwater fishes now listed as threatened or endangered due to alteration and destruction of lotic systems (Moyle and Leidy 1992; Naiman et al. 2002).

Aquatic ecosystems are extremely diverse but are threatened by human landscape alteration. Freshwaters systems, despite not only covering a small proportion of the Earth's surface, are extremely ecologically diverse (Dudgeon et al. 2006) but also some of the most heavily affected by human activity (Malmqvist and Rundle 2002). River regulation has resulted in over half of global freshwater resources being used by humans (Jackson et al. 2001), while more than 50 % of the world's large rivers are fragmented by dams (Nilsson et al. 2005). Streamflow

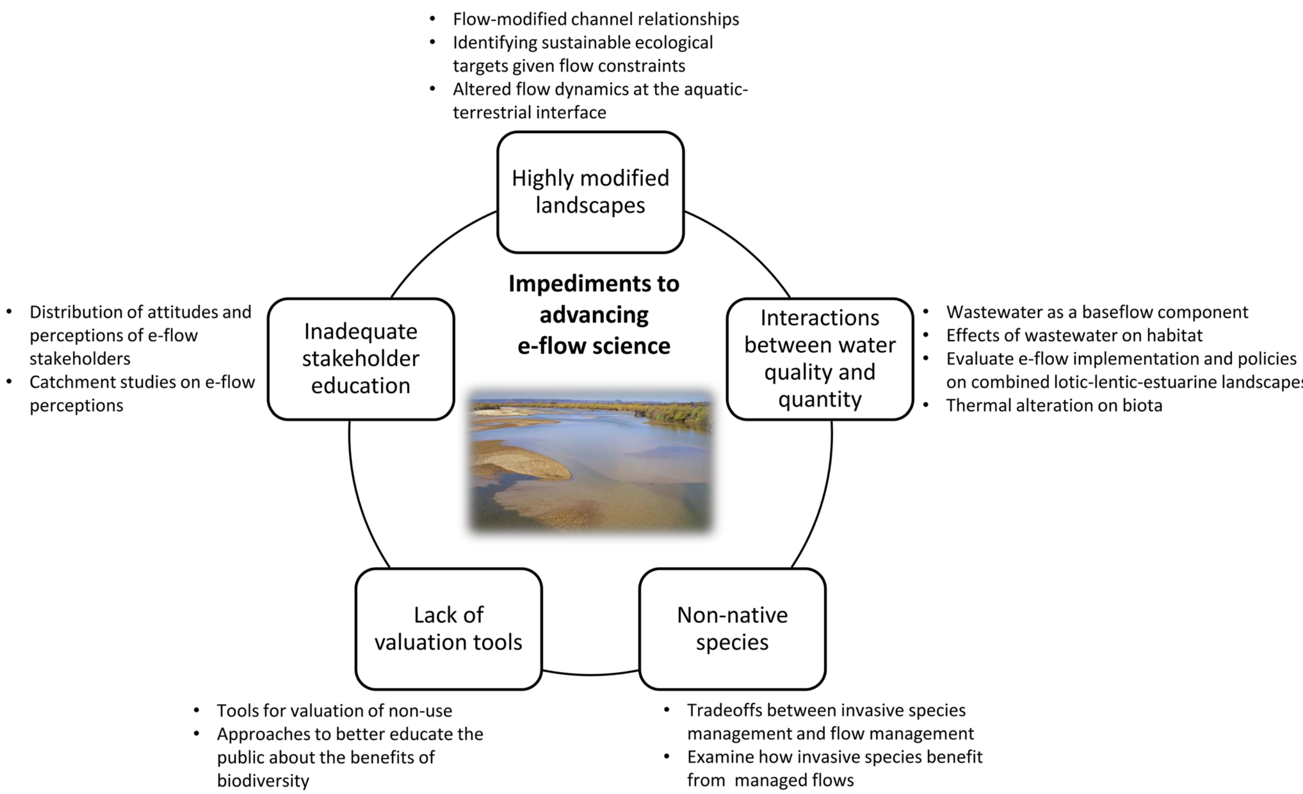

Fig. 1 Major topics where research advances would benefit environmental flow (e-flow) science: existing environmental frameworks and implications for highly modified landscapes, interactions of water quality and quantity, invasive species, identifying the non-use economic values, and the spatial distributions of attitudes and perceptions of environmental flows. The primary research recommendations are listed in bullets next to each topic

alteration to freshwater systems, combined with additional threats, has negatively affected coastal ecosystems (e.g., estuaries, salt marshes, and mangrove forests). Over three billion people (around 40 % of the world's population) live within 100 km of a coastline with growth rates estimated at 16 % per decade (Millennium Ecosystem Assessment 2008). In addition to the direct effects of concentrated human populations, coastal ecosystems are profoundly influenced by damming and other inland anthropogenic activities (Montagna et al. 2002).

Anthropogenic threats to aquatic ecosystems are generally organized into five main categories: overexploitation, pollution, habitat degradation and loss of connectivity, invasive species, and flow modification (see Fig. 1, Dudgeon et al. 2006). The relative importance of these factors is spatially variable and differs among species (Richter et al. 1997). There is also recognition that multiple stressors often interact to affect aquatic organisms (Miller et al. 1989). The legacy of human alteration of freshwater systems is extensive (Downs and Gregory 2004), and although there has been progress in addressing certain issues (e.g., water quality improvements in some developed nations; Allan and Flecker 1993), future changes in climate, population growth, and continued land-use alteration are likely to further exacerbate stress on aquatic ecosystems (Sala et al. 2000). In response to these pressures, freshwater organisms are exhibiting higher extinction rates than their terrestrial counterparts (Ricciardi and Rasmussen 1999; Revenga et al. 2000; Jenkins 2003), with certain species groups disproportionally affected (e.g., Strayer et al. 2004; Dudgeon et al. 2006).

The relationship between flow regime and the aquatic biota is pervasive and multifaceted, to the extent that discharge has been described as a 'master variable' controlling stream ecosystems and driving ecological processes (Fig. 1, Hart and Finelli 1999; Power et al. 1995). Both directly and indirectly, the flow regime affects other regulatory factors such as water quality and temperature thereby regulating ecological integrity (Poff et al. 1997). Discharge is a key driver of physical habitat providing a template determining both taxonomical and functional community structure (Poff and Allan 1995; Bunn and Arthington 2002; Bêche et al. 2006). Disturbance events promote habitat complexity and patchiness through lateral and longitudinal connectivity, and natural fragmentation (Lake 2000). Sediment and solutes are transported through the system creating spatial and temporal gradients in physicochemical conditions. Discharge patterns maintain the river channel (Gordon et al. 2004) and shape the transfer of energy throughout the river system longitudinally (e.g., River Continuum Concept, Vannote et al. 1980), laterally (e.g., Flow Pulse Concept, Junk et al. 1989), and vertically (e.g., Hyporheic Corridor Concept, Stanford and Ward 1993). Discharge also stimulates biotic processes like nutrient uptake and decomposition (Doyle et al. 2005) and predator–prey interactions (Hart and Finelli 1999).

There is an obvious link between aquatic biota and human use via fishing but there are also various non-use activities affected by environmental flow decisions including recreation (e.g., canoeing, birding). In North America, fishing generates $100 billion each year to local economies (American Sportfishing Association 2011) and these economic returns have arguably been one of the most common recreational values quantified relative to environmental flows or water use (Loomis et al. 2003; Katz 2006). In European rivers, the link between high flows, upstream migration (Solomon and Sambrook 2004; Thorstad et al. 2008), and recreational angling catches of Atlantic salmon *Salmo salar* (Alabaster 1970; Aprahamian and Ball 1995) has been highlighted, driving an interest in using artificial water releases to stimulate upstream movement (e.g., Thorstad and Heggberget 1998; Alfredsen et al. 2012). However, many other river recreation activities are growing in popularity with considerable overlap in participation. For example, Nadel (2005) indicated that >82 million USA residents hunted, fished, and viewed wildlife and many of the activities were in combination. In 2003, an estimated 72 million Americans participated in recreation boating activities and over $200 million was spent on the purchase of canoes and kayaks (Nadel 2005). It is anticipated that canoeing alone will increase substantially by 2050 (Bowker et al. 1999). The perceived quality of these activities will be affected by environmental flow designations. Brown et al. (1991) found that of 25 reviewed studies (primarily related to canoeing and kayaking), all found recreational quality increased with discharge until some threshold was reached where quality declined. The thresholds reached varied with activity and experience level of the recreationist. As river and lake recreation increases, these activities may benefit local economies, particularly in rural areas. For example, the development of recreation trails is a form of tourism with substantial economic benefit. The Northern Forest Canoe Trail of New England, USA, attracted 90,000 visitors in 2006 alone and generated approximately 12 million dollars for local economies (Pollock et al. 2012). The growing popularity and value of these activities will no doubt contribute to the increasing complexity of environmental flow decisions, particularly as pressures increase on our limited aquatic resources.

Future changes in climate and population growth will amplify stress on global water resources (Duda and El-Ashry 2000; Sheffield and Wood 2008; Vörösmarty et al. 2010) making environmental flows an important component of management strategies. Environmental flows represent a legal mechanism that is used to balance providing for existing water uses, meeting the needs of future uses, and maintaining aquatic ecosystems and other non-use values (e.g., recreation). Although the legal framework varies geographically, environmental flows are defined as flows necessary to sustain aquatic ecosystems and the 'human livelihoods and well-being that depend on these ecosystems' (Brisbane Declaration 2007). The seemingly simple question of how much water should be left in an aquatic system is compounded by a multitude of societal demands, policies, and needs for advances in our understanding of various ecological interactions. The objective of this paper was to identify the major threats or areas where scientific advancement would provide the most benefit to successful implementation of environmental flows. We identified five major topics where research would be needed to advance environmental flow science: existing environmental frameworks and implications for highly modified landscapes, interactions of water quality and quantity, invasive species, identifying the non-use economic values, and the spatial distributions of attitudes and perceptions of environmental flows (Fig. 1).

Existing Environmental Frameworks

An extensive list of environmental flow frameworks has emerged with the developing discipline of environmental or instream flow science. The Instream Flow Council places the techniques in three basic categories: standard setting, incremental, and monitoring/diagnostic, although some hybrids certainly exist (Annear et al. 2004). Standard setting techniques (e.g., Tennant method) are often policy-driven and set 'rules' to define flow limits (Annear et al. 2004). They are advantageous because of simplicity and cost; however, there are many disadvantages including failure to maintain existing fisheries (Stalnaker 1993; Stalnaker and Wick 2000). Incremental methods (e.g., physical habitat simulation) are site-specific field evaluations of alternative flow regimes. Incremental approaches rely on measurements of river ecohydrological conditions at incremental discharges to analyze river discharge-stage-ecological (biotic and abiotic) relationships (Annear et al. 2004). These methods are some of the most widely applied but can sometimes be costly and time-consuming (Annear et al. 2004) and have resulted in mixed outcomes when evaluated. Gore and Nestler (1988) provide an overview of some common criticisms. Lastly, monitoring methods (e.g., Range of Variability Approach) assess temporal changes in biotic or abiotic conditions related to flow alteration (Annear et al. 2004). The methods vary on the focus of the specific resource component and have been categorized by Annear et al. (2004): hydrology, biology, geomorphology, connectivity, and water quality.

Each of the environmental flow frameworks may be applicable under a variety of societal and geographic settings; however, the relatively recent move toward regional water planning has emphasized a process that includes hydrologic properties, stream classification, and biological responses. This framework is based on the idea accepted by the ecological community that organisms have evolved with the natural flow regime (Poff et al. 1997) and that the relationships between organisms and flow provide a basis for our understanding of how flow alteration will affect the biological community. This led to the development of a subdiscipline focusing on stream classification based on hydrologic properties (e.g., Belmar et al. 2011; Olden et al. 2012; Reidy Liermann et al. 2012).

Streams are classified by similar hydrologic properties as a foundation for applying streamflow recommendations across classes of streams (i.e., negating the need for site-specific flow evaluations unless a follow-up is needed or desired). The ecological limit of flow alteration (Poff et al. 2010) is a holistic planning effort (Richter et al. 2012) that was developed to provide quantitative information to support environmental flows at a regional scale. In addition to stream classification, this approach defines the 'natural flow regime' for the region, assesses the level of alteration to hydrologic conditions, and establishes univariate flow-ecology relationships as a foundation for societal and policy discussions (Poff et al. 2010).

Two recent criticisms of the aforementioned process include: (1) univariate relationships may be insufficient to develop predictable flow-ecology relationships due to other environmental factors (e.g., morphology and temperature; McManamay et al. 2013) and the confounding relationship among multiple flow components (Davis and Brewer 2014) and (2) questionable application to rivers and/or regions that have a highly altered landscape (e.g., Great Plains, Davis and Brewer 2014), and locations where uncertainty is high and physicochemical conditions are extremely unique (e.g., tailwaters; McManamay and Bevelhimer 2013). This leads to the inevitable conclusion that comparing the flow regime to 'natural' conditions may not be appropriate or feasible in many highly altered landscapes of the world and represents an area where future research efforts would benefit river managers. Often, anthropogenic development pre-dates available flow data, requiring modeling approaches to extrapolate from nearby unregulated catchments (e.g., the River Tay, UK, Birkel et al. 2014). Further, in many cases, historical or pre-disturbance flows are not an option, because water volume is limited, overallocated, or is serving multiple purposes (e.g., hydropower and recreation). For example, water allocation in Lake Tenkiller, Oklahoma is over 100 % allocated; therefore, current water law would prevent anything resembling the natural flow regime in the lower river. In addition, the regulatory context may be so complex and multifaceted that recommendations on the basis of the natural flow regime are impractical or target the wrong regulatory entity (e.g., least impactful stressor; Pearsall et al. 2005).

Needs for Highly Altered Landscapes

The science of environmental flows is intended to balance the multidimensional nature of shared water resources in a human-dominated world. Even still, as human population growth continues and water supplies become limited, environmental flow science will be challenged to ensure river flows sustain aquatic ecosystems within highly altered landscapes. In part, this is due to the scale of the problem (i.e., the global extent of hydrologic alteration).

Dramatic alterations to waterbodies and their surrounding landscape have left very few rivers in their natural hydrologic state (Fig. 2). For example, 56 % of stream miles within the USA have habitats that are estimated to be at least moderately degraded and 27 % is estimated to be at least highly degraded (NFHP 2010). River flows have either been altered directly (i.e., channelization and impoundment) or indirectly (i.e., landscape modification). Over 50 % of the global landscape has been transformed or degraded as a result of human activity (Vitousek et al. 1986). Irrigation is the largest consumer of water around the world (Postel et al. 1996); however, other major sources of water loss through land transformation, such as deforestation, are increasingly recognized as significant contributors to the global water cycle (Gordon et al. 2004). Within urban landscapes, extensive impervious surfaces decrease infiltration of precipitation, leading to episodic high storm runoff followed by extremely low, or even intermittent, base flows (Brown et al. 2009). Global estimates of the percent of large rivers regulated by dams range from 50 %, on average, to as high as 77 % for specific regions (Nilsson et al. 2005; Lehner et al. 2011). Large dams tend to stabilize the natural extremities of hydrographs by diminishing flood flows, elevating baseflows, and reducing overall variability; thus, extensive dam regulation is likely to have homogenized regionally distinct streamflow patterns (Poff et al. 2010), potentially leading to losses in continental-wide aquatic biodiversity (Moyle and Mount 2007). Moreover, the effects of flow regulation of river systems are cumulative and certainly affect our estuaries and marine systems (Wilber 1994; Christensen et al. 1998; Livingston et al. 2000). We conclude that hydrologic alterations are widespread and will likely increase in the future regardless of the cause (direct or indirect alteration).

Although the global state of hydrologic alteration substantiates the need for streamflow protection, the ubiquitous nature of highly modified landscapes poses fundamental challenges for environmental flow science. First, determining environmental flow measures, either as recommendations for improvement or as thresholds for protection, is primarily based on our knowledge of natural ecosystem behavior. However, according to the summary provided above, natural ecosystems are becoming rare. More importantly, rivers draining altered landscapes are likely to function and behave differently than natural systems (Brown et al. 2009). For example, peak flood events are periodic, but essential to maintaining floodplain-channel interactions and scouring sediment in natural streams; however, peak floods of similar magnitude may occur more frequently in highly urbanized watersheds and exacerbate poor habitat conditions by scouring limited quantities of wood or gravel substrates (Brown et al. 2009). Potential environmental flow recommendations for urbanized systems would be to dampen or eliminate peak flows and increase baseflows as to artificially stabilize the system. For environmental flow science to adapt, we need to understand the functionality, processes, and pathways for maintaining or restoring some desired degree of ecological integrity in highly altered landscapes.

One of the major challenges of environmental flow protection and enhancement is identifying quantifiable objectives, such as key ecological targets and flow requirements needed to sustain those targets (Richter et al. 2003). These flow requirements may represent adequate ranges of sustainability (Richter 2009) or thresholds, beyond which, ecological impairment is expected (Richter et al. 2012). Out of convenience, the natural flow regime has been used as *the* objective, assuming that this represents both ecological needs and flow requirements. However, identifying ecosystem needs requires intimate knowledge of the dependency of ecosystem function on flows in altered and unaltered systems, and this knowledge gap has been addressed in various settings through workshops based on regional expertise (King et al. 2008). In the absence of such information, Richter et al. (2012) proposed a presumptive standard ranging from 10 to 20 % hydrologic alteration, beyond which ecological functions may be impaired. This is highly suitable for regional planning in areas of limited a priori exploitation; however, it still relies on the natural flow regime as the baseline to measure percent alterations. Highly altered landscapes, however, may already exceed the presumptive standard, as evidenced by a large number of streams with >25 % hydrologic alteration in the USA (Carlisle et al. 2011) and the large areas of Africa, Iberia, eastern Europe, and eastern and south-central Asia 'heavily' or 'severely' affected by river regulation (Grill et al. 2015). Under these conditions, two options exist: (1) either landscape conditions are improved and/or dams are removed or (2) environmental flows must be identified that relate to the sustainability of a few fundamental ecosystem components, indicative of functioning ecosystems. Furthermore, environmental flow prescriptions may not necessarily be informed by the natural flow regime, but be catered to maximize conditions for the target ecosystem components identified, as the natural flow regime may create more harm than benefit. Due to complex stressors and balancing multiple objectives, Bayesian belief networks provide a suitable means to support decision making regarding a few ecosystem components in light of limited information (Webb et al. in press).

Another consideration is that in highly altered landscapes, channel morphology has been modified to such an extent that historical flows may not be beneficial, but potentially harmful (Jackson and Pringle 2010). A recent tendency in environmental flow science has been to recommend flows strictly on the basis of hydrologic records without proper cognizance of the dimensions or morphological condition of the stream channel. Globally, floodplains support a range of human uses (Tockner and Standford 2002); for example, in Europe, approximately 50 % of the population inhabits former floodplain areas (Tockner et al. 2009). To support this human use, channelization and levee construction have disconnected rivers from their floodplain; thus, floods of historical magnitude not only cause destruction of human infrastructure, but also massive losses of limiting habitat features (e.g., large woody debris, Roni et al. in press). In the River Elbe, Germany, reduction in floodplain storage due to urbanization has been linked to increased flood peak water levels at lower discharge in comparison to historic flood events (Becker and Grünewald 2003). River channels of the Great Plains, USA have changed from widespread, braided channels to much more narrow and incised channels that are often disconnected from the floodplains due to bed degradation, upstream dams, encroachment by salt cedar *Tamarix* spp., and adjacent land-use practices. The resulting morphology of the stream channels lacks the capacity to carry historical flow magnitudes (see Fig. 3). Additionally, impoundments trap sediments and bedload; thus, large discharges from dams that exceed transport capacities are likely to eliminate fine substrates unless accompanied by substrate supplementation (Kondolf 1997; Fox et al. 2016). One of the most publicized large-scale restoration projects is the Trinity River Restoration Program, California, a multifaceted restoration to rehabilitate salmon spawning habitats in a diverted stream below Trinity Dam, U.S. Bureau of Reclamation (Trush et al. 2000). A major success has been the implementation of environmental flows that mimic the natural seasonality of the pre-dam flow regime; however, the full magnitude of historical flows has not been implemented due to existing diversions (Trush et al. 2000), but also to avoid flushing valuable substrates downstream (Wilcock et al. 1996b). Furthermore, to ensure adequate salmonid spawning habitat and that the channel would match the new flow regime's capacity, channel reconfiguration and gravel augmentation were also required (Wilcock et al. 1996a). Although regional evaluations of ecological relationships with altered hydrology can assist in developing water policies for states or basins (Poff et al. 2010), knowledge of stream channel-hydraulic interactions are needed when implementing environmental flows within specific streams. Fortunately, a multitude of frameworks are available to aid in this process (Annear et al. 2004).

To date, the field of environmental flows has primarily focused on the water required to sustain ecosystems within individual stream reaches. Although this scope has expanded to consider larger scale dynamics and regional water policies in recent years (e.g., Poff et al. 2010), environmental flow science might benefit from greater interoperability between catchment-scale models (e.g., SWAT) and reach-specific water needs. Constraints on the availability of water to support aquatic ecosystems are likely to stem from strong human demands occurring in the watershed upstream. Environmental flow science will progressively benefit from a synergy between watershed models at basin scales in conjunction with hydraulic modeling of flow-channel dynamics. In addition, environmental flow science has understandably focused on protection and prevention through the development of policies, with most proactive restoration approaches limited to regulatory contexts supported by legislation, such as dam relicensing. Given the widespread extent of altered landscape conditions, coarse-scale research that identifies opportunities to maximize flow enhancement and restoration would be beneficial.

Interactions of Water quality and Quantity

Water quality and quantity are among the key components of environmental flows (Acreman and Ferguson 2010), and have considerable, often interactive, effects on aquatic systems. Although flow magnitude is just one component of the natural flow regime (Poff et al. 1997), the resulting effects on water quality and other processes have implications for ecological integrity (see Fig. 1, Poff et al. 1997). Existing frameworks for understanding effects of environmental flows at local (e.g., PHABSIM, Milhous and Waddle 2012) to regional (e.g., ecological limits of hydrologic alteration, Poff et al. 2010) scales (see also Tharme 2003) may not be suitable in all locations (e.g., tailwaters, McManamay and Bevelhimer 2013).

In arid and highly populated regions, to name a few, baseflow conditions may be sustained via treated sewage effluent. The wastewater discharged into these streams may support ecosystems that would otherwise not exist because of water limitations (Luthy et al. 2015). However, relying on effluent to support these ecosystems can have serious consequences. It is not unusual for polychlorinated biphenyls, polyaromatic hydrocarbons, pesticides, pharmaceuticals, personal care products, metals, and radionuclides to be present following treatment of sewage (Hughes et al. 2014). Wastewater discharges often contain constituents that have the potential to alter fish behavior (e.g., Fluoxetine, fathead minnow *Pimephales promelas*, Weinberger and Klaper 2014), reproductive fitness (e.g., steroidal estrogens, common roach *Rutilus rutilus*, Hamilton et al. 2014) and larval survival and juvenile production (e.g., steroid estrogen, fathead minnow, Schwindt et al. 2014). Wastewater reuse solutions to water shortages (i.e., reclamation of wastewater for repeated irrigation and industrial use) are increasingly used to conserve water resources (Levine and Asano 2004). While these practices augment water supplies and may reduce pollution, they pose alternative threats for lotic systems where treated wastewater may be the primary source of discharge during portions of the year. In Israel where almost all water is allocated for human uses, recycling of wastewater provides a mechanism for river recovery (Juanico and Friedler 1999; Tal 2006). However, a wastewater-derived baseflow reduces a river's capacity to absorb further pollution loading (Juanico and Friedler 1999). In the Canadian River, Oklahoma, wastewater discharge currently supplies much of the baseflow during the summer months and is one of the last remaining strongholds for the federally threatened Arkansas River shiner *Notropis girardi* (Worthington et al. 2014). Major declines in baseflow lead to the loss of longitudinal connectivity that over time has important consequences for the persistence of many riverine populations (Bunn and Arthington 2002). This effect is particularly evident with repeated stream drying in fragmented stream systems of the Great Plains (Perkin et al. 2014). Decreased flows in arid streams exacerbate the influence of non-native predators on fish populations, perhaps by creating less favorable habitat conditions for flow-adapted native species, though the possible mechanisms remain unclear (Propst et al. 2008). Lastly, diminished baseflows change the dynamic between surface water and groundwater interactions (Hynes 1983; Gordon et al. 2004), especially for hyporheic flows that create thermal refugia for fish populations (Power et al. 1999; Sutton et al. 2007). Loss of these refugia created via groundwater interactions has been related to the decline of several fishes (e.g., Arkansas Darter, *Etheostoma cragini*, Eberle and Stark 2000).

Water quality includes chemical and thermal characteristics and is an important but often neglected component of environmental flows (Nilsson and Renöfält 2008; Olden and Naiman 2010). Decreased water quality is one of many consequences of human-altered flow regimes (Abromovitz 1996; Collier et al. 1996). Water quantity reductions concentrate solutes, leading to increased salinity (Bradley et al. 1990) among other constituents. High pulse flows are capable of flushing waste and pollutants following prolonged low-flow periods (Postel and Richter 2003). Heating capacity, and thus stream temperature, is inherently linked to the volume of water in a system (Caissie 2006). Low flows can lead to more frequent high-temperature events, which can result in reproductive stress in freshwater mussels (Galbraith and Vaughn 2009), and exceed critical temperatures for fishes (Sinokrot and Gulliver 2000) leading to bioenergetic loss (Elliott 1976) or even death (Fry 1947; Brett 1956). Despite the effects of water temperature on ecological patterns, the thermal regime has been insufficiently considered in the context of environmental flows (Olden and Naiman 2010). The thermal effects of dams contribute to the unpredictability of tailwater systems and represent a challenge for environmental flow assessments, particularly given the lack of suitable frameworks for these ecosystems.

The location of water released from a dam or diversion alters the thermal regime thereby affecting downstream biota. Failure to consider temperature effects of dam releases may lead to failure to replicate natural ecological processes (King et al. 1998). Altered thermal gradients below dams can extend for hundreds of kilometers downstream (Ellis and Jones 2013). Increased water temperatures below dams have led to accelerated migrations and earlier spawning of sockeye salmon *Oncorhynchus nerka* in the Pacific Northwest (Quinn et al. 1997), while hypolimnetic cool water releases resulted in reduced growth, later smoltification and lower production of Atlantic salmon and anadromous brown trout *Salmo trutta* in the River Surna, Norway (Saltveit 1990). Temperature gradients caused by dam releases also influence spatial patterns of gizzard shad *Dorosoma cepedianum* spawning in the Savannah River, between Georgia and South Carolina (Paller and Saul 1996). Alternatively, decreases in stream temperatures may extirpate native species from these locations (Haxton and Findlay 2008; Olden and Naiman 2010). For example, hypolimnetic releases have led to the development of tailwater trout fisheries below many dams in systems that were naturally considered to be 'warmwater' (e.g., lower Illinois River, Oklahoma, Guadalupe River below Canyon Dam, Texas). Temperature, however, is not the only physicochemical parameter altered downstream of dams and diversions.

Water releases may create additional water quality constraints in tailwater systems (i.e., the stream segment downstream of a dam). Hypolimnetic releases often result in downstream anoxia and it can be particularly difficult to engineer structures at the dam that may improve these conditions (e.g., lower Illinois River, Oklahoma). In addition, water releases are still necessary to improve downstream conditions, often a source of political and economic debate, particularly where reservoir waters are over 100 % allocated. Improvements in water quality at these locations will need to address water quantity using a framework that differs from traditional environmental flow approaches (e.g., water quality analysis simulation program modeling, Y. Zhou, unpublished data).

The interaction of reduced water quality and quantity has also severely altered the integrity of coastal ecosystems worldwide. Coastal ecosystems are unique brackish environments that form an interface between freshwater and marine environments. The ecosystem services provided to humans by coastal habitats include high productivity (Millennium Ecosystem Assessment 2008), nutrient cycling (Basset et al. 2013), reduced magnitude of flood events (Temmerman et al. 2013), and diminished intensity of storm surges (Barbier and Enchelmeyer 2014). Coastal ecosystems often also exhibit high species diversity (Basset et al. 2013), harbor endemic biota (Marrack et al. 2015), and form important nursery habitat for fish species (Bertelli and Unsworth 2014). Coastal habitats are among the most threatened ecosystems on the planet. For example, at least 35 % of mangrove forests worldwide have been destroyed (Millennium Ecosystem Assessment 2008). In addition to the large human populations in coastal areas that have affected ecosystem dynamics through excessive use of freshwater and changes in land-use practices (Montagna et al. 2002; Millennium Ecosystem Assessment 2008), inland human activities can also impair coastal habitats. Nutrients and pesticides from intensive agricultural activities that enter inland streams and rivers through runoff ultimately reach coastal areas, leading to reduced water quality with detrimental effects to biota. For example, nitrogen loading in coastal ecosystems has doubled worldwide, which has caused severe eutrophication (Millennium Ecosystem Assessment 2008) and large hypoxic zones (Ritter and Montagna 1999; Montagna and Froeschke 2009). Alternatively, coastal ecosystems also rely on inlands flows for the delivery of both freshwater and sediment. Damming and other diversions have decreased streamflow delivery rates to coastal areas by 30 % worldwide (Millennium Ecosystem Assessment 2008). Reduced freshwater inflow to coastal ecosystems alters their unique environmental characteristics that form the freshwater-marine transition. The reduction of freshwater delivery to coastal habitats can reduce ecosystem metabolism, thus lowering productivity (Russell et al. 2006). Furthermore, reduced freshwater inflow to coastal ecosystems has shifted the salinity gradient with more saline waters found further inland (Millennium Ecosystem Assessment 2008). Few brackish water organisms are tolerant of high salinity and the inland intrusion of seawater can profoundly alter the structure of coastal habitat communities (Sklar and Browder 1998). The large spatial scales that influence coastal ecosystems present unique challenges for both research and policy development (Montagna et al. 2002).

Because coastal habitats are heavily influenced by upland activities, proper ecosystem functioning requires effective environmental flow standards across large spatial scales. Increased awareness of a global hydrologic cycle has led to the realization that environmental flow plans need to be extended from local to regional scales. Some states in coastal areas have established environmental flow standards (e.g., Texas, Opdyke et al. 2014); however, the development of multi-state environment flow policies remains in its infancy and data to establish flow-ecology relationships across broad scales are severely lacking (Poff and Matthews 2013). In addition, strategies to increase freshwater inflow rates to coastal ecosystems must accommodate human needs to gain political support (Montagna et al. 2002). Researchers have proposed flexible broad-scale environmental flow plans that can be implemented in a manner that will meet ecological objectives while avoiding economic loss and land-use conflicts (e.g., Overton et al. 2014; Pang et al. 2014; Yang and Yang 2014), including plans catered to coastal habitats. For example, Sun et al. (2015) proposed an environmental flow framework that examines spatial and temporal patterns of inland streamflow on estuarine habitats while considering human land-use requirements. Unfortunately, widespread implementation and evaluation of proposed environmental flow standards remain a major hurdle (Davies et al. 2014). An additional challenge for developing effective environmental flow standards for coastal ecosystems is their unique environmental characteristics. Since they form the freshwater-marine interface and are influenced by both ocean and inland flow patterns, the ecological response to freshwater delivery is complex and non-linear in nature (Montagna et al. 2002). Simply increasing freshwater delivery rates will not restore proper ecosystem functioning of coastal habitats without consideration of other elements of the flow regime (e.g., timing and duration of flows, Koster et al. 2016).

Increasing water demands require innovative solutions to balance human needs with ecological needs, given the interplay between water quantity and quality. Targeted studies to understand potential enhancement of water quality while maintaining water quantity and meeting human demands have shown promise. For example, methods have been developed to improve water quality below dams with either minimal (Krause et al. 2005) or beneficial (Hayes et al. 1998) effects on hydropower operations. Managing for multiple goals in tailwater systems (e.g., sport fisheries, biodiversity, and habitat enhancement) is another issue of interest. How do we balance human water demands with conservation of imperiled species, particularly in water-stressed regions? Ultimately, understanding which components of water quality and quantity are manageable will be essential to determining appropriate trade-offs for meeting human and ecosystem needs. More importantly, policy development across political boundaries will be necessary to ensure coastal regions and some interstate large river systems benefit from environmental flow standards.

Invasive Species

The link between flow alteration and changes in aquatic communities is well established (Bunn and Arthington 2002; Poff and Zimmerman 2010 and references therein), although the magnitude and direction of the response are often difficult to predict (see section above on existing environmental frameworks). A modified flow regime often allows for the establishment of non-native aquatic and riparian species (Stanford et al. 1996; Bunn and Arthington 2002; Poff and Zimmerman 2010). This operates across a continuum with successful establishment of non-native species greatest in systems with the highest levels of human disturbance (Moyle and Light 1996a, b). This has resulted in several authors suggesting restoration of the natural flow regime as a mechanism for safeguarding native species while excluding introduced species (Poff et al. 1997; Marchetti and Moyle 2001; Brown and Ford 2002; Lytle and Poff 2004). However, it is generally accepted that flow regime manipulation in isolation is not a panacea for the restoration and conservation of native aquatic species (Saunders et al. 1998; Englund and Filbert 1999; Propst et al. 2008).

The multitude of threats to freshwater biodiversity rarely operates independently (Fig. 1, Dudgeon et al. 2006). For example, the spread of *Tamarix* species in the USA has been linked to the reduction of high-flow events following river regulation, once established *Tamarix* stands drive sediment aggregation and channel narrowing further altering the hydrological regime and instream habitat (see Tickner et al. 2001). Therefore, feedbacks between individual threats may result in potential unintended consequences from restoration, such as restored flow conditions benefiting non-native species (Marks et al. 2010). Although examples of flow regime restoration leading to the reestablishment of native species are apparent in the literature (e.g., Scoppettone et al. 2005; Kiernan et al. 2012), both positive and negative ecological feedbacks have been observed and predicted (Reich et al. 2010). The restoration of a more natural flow regime in New Zealand streams is predicted to allow non-native brown trout *Salmo trutta* to colonize areas that provide refuges for the native roundhead galaxias *Galaxias anomalus* potentially leading to the species' extirpation (Leprieur et al. 2006). Likewise, increased lateral connectivity in the Willamette River, Oregon due to re-establishment of high flows is predicted to negatively affect Oregon chub *Oregon ichthys crameri* by increasing dispersal of non-native species into isolated floodplain populations (Scheerer 2002). In some circumstances, the management of non-native species may oppose the goals of restoring of the natural flow regime. Intentional fragmentation has been used as a mechanism to stop the expansion of several non-native species (see Table 1 in Rahel 2013). For example, within Iowa certain dams could be retained to provide a barrier to the upstream movement of Asian carp (e.g., big head carp, *Hypophthalmichthys nobilis* and silver carp, *Hypophthalmichthys molitrix*) (Hoogeveen 2010), despite restoration of long free-flowing reaches with naturally variable hydrologic regimes being highlighted for the conservation of native minnow species in the wider Missouri River basin (e.g., Dieterman and Galat 2004; Quist et al. 2004). Our challenge moving forward will be to maintain or increase hydrological connectivity while minimizing the spread of certain taxa (Rahel 2013).

Biotic responses to restoring the natural flow regime are dependent on the interaction between hydrology, geomorphology, and ecological traits of the in situ species (Shafroth et al. 2010). In the San Juan River of Colorado, New Mexico, and Utah, reservoir releases designed to mimic historic spring discharge resulted in little change in the abundance of introduced species particularly among small-bodied, short-lived, fecund taxa (Propst and Gido 2004). For management projects, the particular flow regime components modified (e.g., magnitude, timing) are critical to the ecological response. In the Colorado River, Arizona, high magnitude, short duration flood pulses released from the Glen Canyon dam in March and November limited *Tamarix* establishment; conversely, a smaller, prolonged event in May followed by a steady summer discharge, designed to increase humpback chub *Gila cypha* recruitment, produced widespread *Tamarix* establishment as the flow coincided with the species' seed release (Mortenson et al. 2012). Likewise, the timing of managed flood inundation in Macquarie Marshes, Australia was implicated in stimulating recruitment of non-native species at the expense of native fauna (Rayner et al. 2015). It is pertinent that prior to undertaking flow manipulation, managers have a firm understanding of the phenology and flow requirements of both native and introduced species to determine the potential implications of restoration actions. This is highlighted by the fact that few studies have weighed the relative contributions of flow regime restoration and non-native species removal to the conservation of native biodiversity. However, Marks et al. (2010) demonstrated that flow restoration had a smaller effect on native fish species abundance in comparison to removal of non-native species, which was achieved for a greatly reduced economic cost. The authors suggest that in areas where non-native species are still established flow restoration had no effect (Marks et al. 2010). To both minimize the persistence and spread of non-native species, while promoting conservation of native taxa, a structured approach that considers the likely response of all species within the community to flow regime restoration is warranted. In many areas, such an approach may be limited by a lack of basic knowledge of species' ecological and life-history requirements in relation to flow.

Another concern of invasive species as a component of environmental flows is evaluating their effects on hydrological alteration, especially related to their growth forms and evapotranspiration potentials. For example, giant cane *Arundo donax* transpires 69.3 million m^3 of water per year on the Santa Ana River, California compared to an estimated 23.1 million m^3 that would be consumed by native vegetation (Giessow et al. 2011). Giant cane is expansive across the lower Rio Grande Valley, Texas and along with water hyacinth *Eichhornia crassipes* and hydrilla *Hydrilla verticillata* contributed to the Rio Grande River, Texas ceasing to flow to the Gulf in 2001 (Rister et al. 2011). Further, in the Mar Chiquita coastal lagoon, Argentina, the presence of the non-native reef-building polychaete *Ficopomatus enigmaticus* altered both the sediment transport and water flow in the lagoon (Schwindt et al. 2004).

Non-use Economic Value

Although the capitalist economic system encourages the overexploitation of natural resources ('tragedy of the commons'; Hardin 1968), monetary value can still be derived by conserving 'nature.' In fact, the economic value of ecosystem services may equal or exceed the world's gross domestic product (Costanza et al. 2014). At the local scale, the conservation of natural resources may provide economic benefits that exceed 100 times the cost (Balmford et al. 2002) and an investment in biodiversity can promote economic growth via tourism (Freytag and Vietze 2013). For example, birdwatching (a completely non-consumptive activity) generates over $100 billion in revenue each year in the United States (USFWS 2013). Recreational fishing also provides over $100 billion each year to state and local economies through sales of licenses, equipment, bait, food, etc. (American Sportfishing Association 2011). Sustainable recreational fishing ensures economic benefit in future years, thus providing a tangible value for environmental flows. For example, fishing for Guadalupe bass, *Micropterus treculii* in the Edwards Plateau ecoregion, Texas, USA generates over $75 million and 776 jobs in a sparsely populated region (Thomas et al. 2014). Likewise, angling for Atlantic salmon and sea trout in England and Wales attracts visitors to rural areas, supporting jobs and contributing to the local economy (Peirson et al. 2001; Aprahamian et al. 2010). However, game fishes make up only a small percentage of freshwater fish species and recreational fishing qualifies as a use of water resources (albeit much less exploitative than commercial fishing). The lack of 'charismatic megafauna' among freshwater biota makes conservation efforts challenging for non-game species. Therefore, improved public support of environmental flows requires economic value to be associated with ecosystem services provided by aquatic organisms and their habitats. Meeting these challenges require an effective collaboration of ecologists, social scientists, and economists. Two interrelated areas that need to be addressed to facilitate the implementation of environment flows are (1) better educating the general public about the value of biodiversity and (2) accurate economic valuations of non-use.

Ecosystem services are benefits provided to people by nature through direct functions and/or esthetics. Freshwater fishes provide numerous benefits to the general well-being of humans through nutrient cycling, algae control, and food web stability (see Holmlund and Hammer 1999). Since aquatic vegetation is often seasonal or ephemeral, fish are often analogous to trees in freshwater systems by playing an important role in long-term nutrient storage (Allen and Hoekstra 1992). Aquatic plants and invertebrates also provide ecosystem services via improved water quality (de Bello et al. 2010). However, the long-term advantages to human prosperity through the conservation of biodiversity are largely (almost exclusively) overlooked when installing impoundments or increasing water withdrawals. Despite an inherent economic importance, ecosystem services can only be valued through knowledge. Individuals with expertise in outreach are essential to providing the general public with a basic, yet holistic, understanding of freshwater ecosystems. Better awareness of the 'big picture' will not only increase public support, but will also improve the effectiveness of non-use valuation methods through better-educated responses to surveys. The Environmental Protection Agency's (EPA) Healthy Watershed Initiative is an example of a proactive approach incorporating ecosystem services, public health and stakeholder involvement for protecting watersheds and their aquatic resources (EPA 2016).

An accurate valuation of the non-use of natural resources is challenging through conventional economic assessments. Kahn (2005) outlines three methods to assess the value of non-market goods. (1) Revealed preference techniques provide a value of an environmental amenity through market value. For example, if two properties are identical with the exception of one being located near a pristine stream and the other a degraded stream, the difference in market value of the properties reveals a value of water quality. (2) Benefit transfer techniques use values from other studies and apply them to a current problem. For instance, economic losses in one area resulting from decreased fishing due to decreased water quality can be used to project losses in another area. (3) Stated preference techniques use surveys to elicit an individual's willingness to pay for a stated environmental quality. The most common stated preference for valuing ecosystem services is contingent valuation. For example, Hejazi et al. (2013) determined that users of a freeway would pay $77 million annually (mean of $1.84 per household) to conserve natural resources along the route (primarily for esthetic value). Loomis et al. (2000) used contingent valuation to determine that households living along the Platte River in Colorado would cumulatively contribute at least $19 million annually to restoration efforts. Contingent valuation methods have been used to evaluate the non-market value of both wetlands (e.g., Brouwer et al. 1999; Grammatikopoulou and Olsen 2013) and streams (e.g., Brown and Duffield 1995; Berrens et al. 2000; Barak and Katz 2015). Contingent valuation can also be used to value water for recreation versus consumption (e.g., Loomis 2012; Loomis and McTernan 2014). The accuracy of contingent valuation is often criticized and requires skilled economists to implement effective surveys (Kahn 2005). In particular, the value of threatened and endangered species appears to have increased over time but we need more consistent measures of those values to avoid criticism (Richardson and Loomis 2009). Nevertheless, contingent valuation allows a monetary value (i.e., a number) to be associated with non-market goods such as environmental flows (Diamond and Hausman 1994; Franco and Luiselli 2013).

Human Dimensions

Traditionally, studies addressing human perceptions of the natural environment assumed that responses are distributed evenly across the landscape of interest; however, more recent evidence has suggested that environmental concerns or perceptions by people represent clustered patterns on the

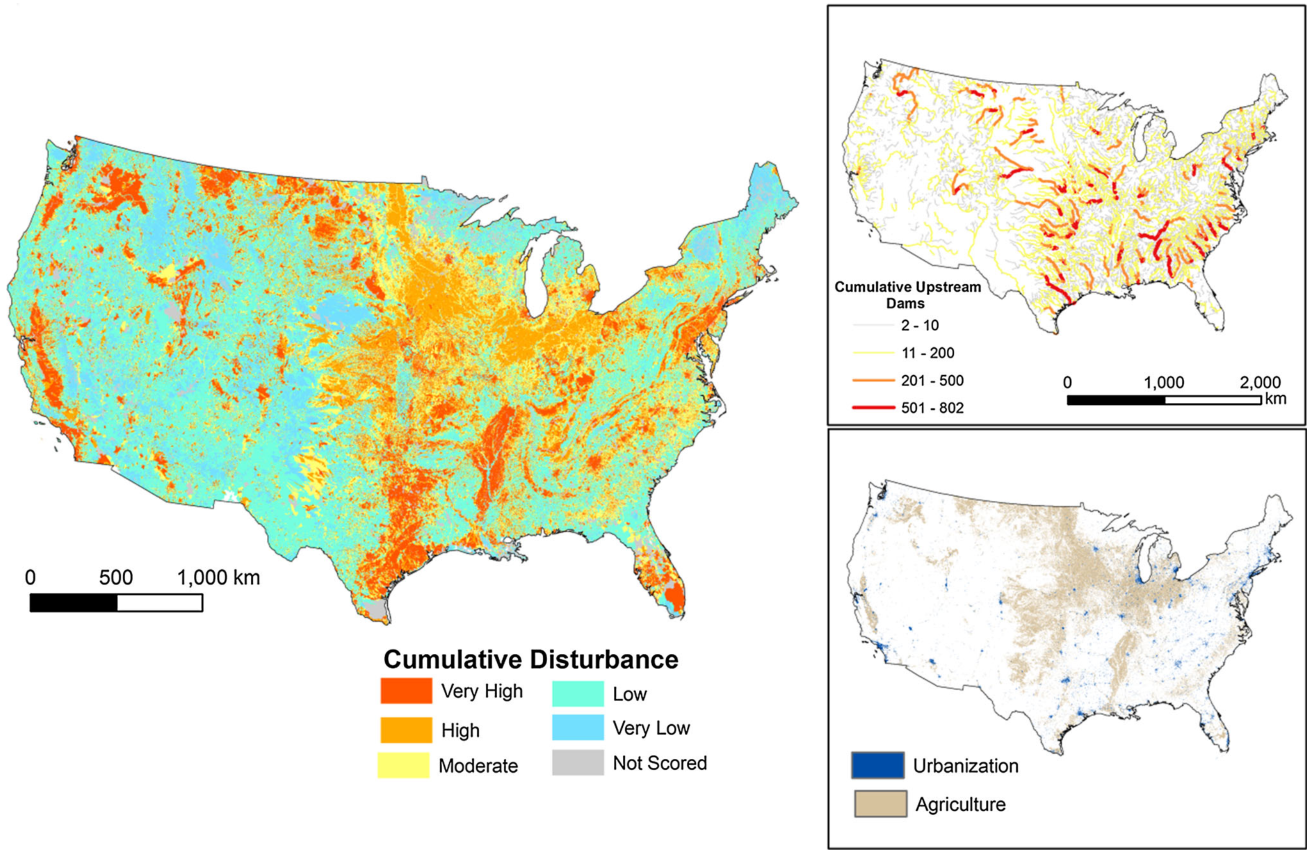

Fig. 2 Altered landscape conditions in the conterminous USA. Cumulative disturbance index represents potential risk to fish habitats (NFHP 2015). The *right inset* depicts two major factors contributing to the index: cumulative number of upstream dams from NFHP (2015) (*top inset*) and urban and agricultural landscapes from MRLC (2016) (*bottom inset*)

landscape. These spatially correlated patterns were influenced by a variety of factors including population density, demographics, socioeconomics (Brody et al. 2005), and proximity to the environmental concern in question (Brody et al. 2004). For example, there was a positive relationship between the income of farmers and their attitude toward ecosystem services (Poppenborg and Koellner 2013). Brody et al. (2004) found perceptions of water quality to relate to driving distance to the water body. These results have significant policy implications in that new water policies would be expected to be perceived differently by the community depending on the perceptions of the first stakeholder groups engaged (Brody et al. 2005). Understanding the distribution of attitudes and perceptions of water policies appears to be an important prerequisite to gaining support for advancing environmental flow science and for successfully implementing new environmental policies. Underlying attitudes and perceptions will also provide some insight to the knowledge base of different constituencies and sectors; increasing research shows natural resource/environmental literacy of the public is low (Koyle 2005).

Some of the spatial variation in people's perception and viewpoint of the environment is related to regional dissimilarities in water law and climate. For example, in the more arid western USA, the doctrine of prior appropriation assigns priorities to water rights based on the timing of permit execution and must meet the definition of 'beneficial uses.' Traditionally, 'beneficial uses' required water to be diverted out of the stream channel which prevented environmental flows from becoming established in many areas (Clayton 2009). However, many states have expanded the meaning of 'beneficial use' to include the environment and have relaxed the diversion requirement (Schempp 2009). In areas where overallocation of resources has occurred, buying back water for the environment has been the most acceptable method of reallocating water, particularly in the USA and Australia (Lane-Miller et al. 2013). In the more humid eastern USA, riparian rights allow for 'reasonable water use' to adjacent landowners but also provide designations for instream uses in navigable waters such as swimming, boating, and fishing. However, even in these regions of greater precipitation, the attitudes associated with environmental flows and associated

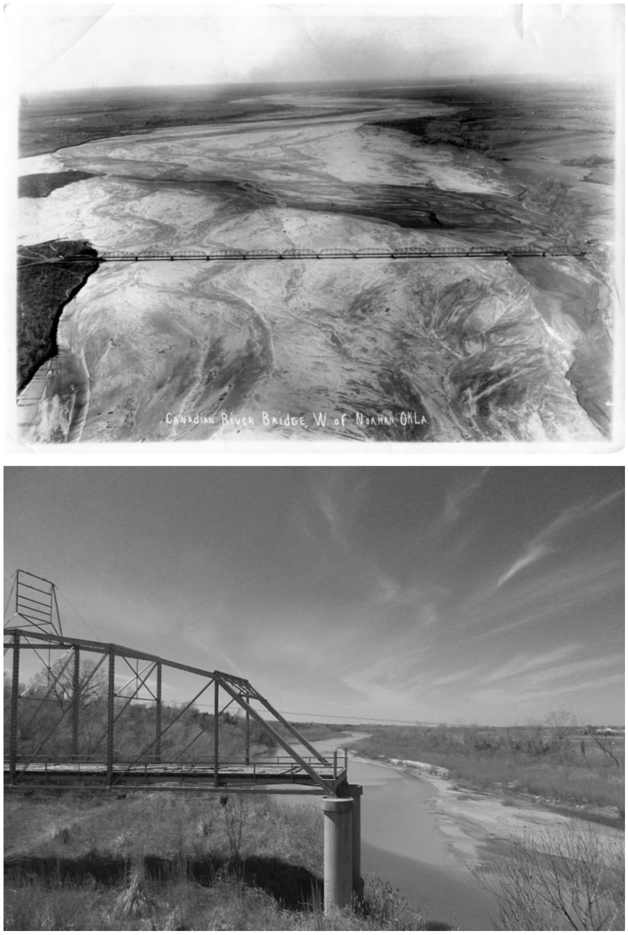

Fig. 3 Photograph comparison of the Canadian River (west of Norman, Oklahoma, USA) in the 1920's (*top panel*) and the current channel (2015) near the same location (*bottom panel*, N 37° 45.017′ W 093° 15.984′). The photograph in the *upper panel* was reproduced with permission from the Research Division of the Oklahoma Historical Society

pressures on existing policy may diverge during times of drought (Loch et al. 2010). However, our knowledge of patch distributions of changing attitudes and perceptions is limited, making it difficult to move policies forward in some watersheds or regions.

Certain geographic factors have been well recognized in determining environmental perceptions (e.g., urban versus rural environments, Tremblay and Dunlap 1978) though temporal changes in these trends have also been acknowledged (e.g., Fortmann and Kusel 1990). Spatial distributions of changing perceptions, particularly in individual catchments, have been much less studied. However, Brody et al. (2005) studied the perceptions and drivers of community views of water quality in small catchments in south Texas and found four distinct groups of perceptions. It is unclear what spatial scale is most appropriate for examining changing perceptions of water quantity and how that scale might align with other disciplines and existing frameworks associated with environmental flows. However, because of the increase in geospatial data and spatial modeling capabilities, it is reasonable to incorporate these approaches into the social sciences and integrate these results with other environmental flow components (e.g., ecology, hydrology, and economics). Understanding the spatial changes in values, perceptions, and attitudes of environmental flows may assist in better development of the ecological questions (e.g., organism focus) and how and when stakeholders are integrated into the process of environmental flow planning. Lastly, as with non-use, the role of social scientists may be important to reduce spatial variation in attitudes and perceptions by better educating the general public about current and future environmental issues.

Conclusions

The future of environmental flow science will rely heavily on collaboration across disciplines to move forward with the aforementioned challenges. Environmental flow science will benefit from this collaboration by (1) providing a better understanding of the ecological needs of native species to offset the often unintended consequences of benefiting non-native species, (2) developing approaches to address streamflow needs in highly modified landscapes where historic flows do not provide reasonable comparisons, (3) integrating water quality needs where interactions are apparent with quantity but not necessarily the proximate factor of the ecological degradation, and (4) providing more obvious ecological study targets and locations by understanding the attitudes and perceptions of local stakeholders. Successful implementation of environmental flow policies will benefit from increasing our understanding of the spatial distribution of attitudes and perceptions across the landscape so that we can better engage the correct stakeholders in the appropriate spatial context. Further, improving our understanding of the non-use economic value will create a metric of comparison for natural resources that will (1) better appeal to city and regional developers and (2) allow for targeted implementation of objectives in regions with highly modified landscapes (e.g., developing environmental flows for persistence of remnant fish populations between reservoirs). We recognize that the interdisciplinary collaboration required to move forward will be difficult (see NRC 2004); however, scientists working in environmental flow science already conduct interdisciplinary studies (e.g., ecological limits of hydrologic alteration). The research momentum could be redirected to address additional components (i.e., altered landscapes with no potential for historic flows) and better integrate the socioeconomic disciplines. Combined, we believe these efforts would increase the overall success of environmental flow science and implementation.

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