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Upstream Landscape Dynamics of US National Parks with Implications for Water Quality and Watershed Management

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1. Introduction

The mission of the United States (US) National Park Service (NPS) is to "conserve the scenery and the natural and historic objects and the wild life therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations" (NPS, 1916). NPS currently manages 397 parks covering about 358,200 km^2, or approximately 4% of all US states and territories. The National Park system includes approximately 300 parks that are considered to contain significant natural resources. These parks are key components of a larger network of protected areas that anchor the conservation of natural resources in the US. They also afford direct protection for a number of important and defining resources in the US, including 421 species of threatened or endangered plants and animals, nearly two-thirds of native fishes in the 50 states (Lawrence *et al.*, 2011), the highest point in North America (Mt. McKinley in Denali National Park, 6,194 m), the longest cave system in the world (Mammoth Cave National Park with more than 587 mapped km of caves), the country's deepest lake (Crater Lake in Crater Lake National Park, 589 m), the lowest terrestrial point in the western hemisphere (Badwater Basin in Death Valley National Park at 86 m below sea level), and – within these extremes – many other natural resources that are significant at local, regional, and national scales.

While protected areas are foundational to a strong natural resource conservation network, ecologists have long recognized that virtually all parks are too small to be self-sustaining ecosystems, and activities outside park boundaries can profoundly impact park resources (Newmark, 1985; US General Accounting Office, 1994; Parks & Harcourt, 2002; Wiersma *et al.*, 2004; Hansen & DeFries, 2007; Hansen *et al.*, 2011). Chief among these activities is the intensification of land use and the appropriation of ecological services. Land use intensification leads to the conversion of natural habitat, which generally results in an overall loss of habitat, fragmentation of remaining natural areas, increases in edge zones, changes in the runoff of water, sediments, and nutrients, and follow-on modification of physical and ecological processes in terrestrial and aquatic ecosystems. Depending on the location, extent, and magnitude of these anthropogenic changes, the effects may propagate over very large areas and have important consequences for resource management in protected areas.

While the NPS mission is to protect all natural resources, water is perhaps the most universally important resource to parks and to protected areas worldwide. Provision of fresh water is a key ecosystem service provided by many parks, and wetlands and riparian habitats are often biological ‘hot spots’ that support disproportionately high levels of biodiversity (Stein *et al*., 2000; Scott *et al*., 2001). Because fresh water resources are so important to parks, the focus of this chapter is on landscape-scale factors that affect water resources and associated values. Flowing water directly connects water resources inside and outside park boundaries. Landscape-scale activities beyond park boundaries can particularly affect water resources and the ability of parks to manage and protect these resources. A means to identify and quantify imposing threats is thus important to designing and implementing effective park management strategies.

To manage an extensive network of protected areas like the NPS system of parks, there is a clear need to assess the system-wide context and status of parks relative to their goals (Scott *et al*., 2001; Svancara *et al*., 2005). Results from broad-scale analyses can identify patterns and trends that may be undetectable at the individual unit scale, and provide guidance for changes in regional or national-level policy. Scott *et al*., (2001), for example, noted that US protected areas (parks, refuges, etc) disproportionately represent lands characterized by high elevation, low productivity, and low rainfall – the places that are cold, dry, and barren. Areas in highly productive river valleys – the location of many biodiversity hot spots – were disproportionately under-represented in the network of protected areas. The widespread availability of broad-coverage, geospatial data on environmental conditions and landscape attributes has facilitated new and sophisticated analyses of the geographical context and anthropogenic impacts to terrestrial, freshwater, coastal, and marine ecosystems at regional, national, and global scales (Sanderson *et al*., 2002; Halpern *et al*., 2007, 2008; Leu *et al*., 2008; Woolmer *et al*., 2008; Lawrence *et al*., 2011).

While a few studies have measured and assessed the landscape characteristics of US National Parks (Scott *et al*., 2001; Svancara *et al*., 2009; Davis & Hansen, in press; Wade *et al*., 2011), these efforts focused on the broader landscape context or specific components of the landscape, rather than watersheds, even though water is one of the most defining resources for parks and other protected areas (Dixon & Sherman, 1991; Hawkins *et al*., 2003). To our knowledge, only Lawrence *et al*., (2011) have rigorously evaluated system-wide the upstream landscape dynamics of US National Parks, specifically from the perspective of maintaining protection for freshwater fish diversity. They used broad-scale datasets to assess both threats to the use of parks as ‘freshwater protected areas’ and the potential capacity to manage activities in the contributing watersheds. Based on a relatively simple set of analyses, but involving computationally intensive operations, they were able to identify a subset of parks that could serve as the foundation for a system that would likely preserve a large proportion of US freshwater fish.

To guide the analyses in this chapter, we asked a series of questions:

1. Based on established ecological principles and landscape-scale data, what is the general context of park upstream watersheds?

2. Which major landscape factors explain most among-park variation in upstream watershed context?
3. What can we infer about the condition of park freshwater resources, and how do these vary geographically?
4. What are the major challenges and opportunities for managing park upstream watersheds?

We first describe the ecological foundation and general approach to evaluating park upstream watersheds. We proceed to describe the selection of variables and data sources used in the analyses, and briefly review the ecological basis for including those variables. We then outline the analytical techniques and criteria for including or omitting parks from the study. The final sections of the chapter present the results of our watershed and water quality analyses, and our interpretation of these results. We conclude with a summary of our principal findings and recommendations for future research.

2. Assessing watershed condition

2.1 Ecological foundation

Figure 1 illustrates our overall conceptual model for assessing parks at a landscape scale. The model acknowledges key anthropogenic drivers (or stressors), important attributes of the natural landscape, and contextual elements that affect conservation and management actions. Analyses that consider these elements can evaluate geospatially explicit broad-scale vulnerabilities and opportunities for conservation and management. Our model is founded on more comprehensive analyses by Hansen & DeFries, (2007) of the mechanisms that link land use intensification to the resources within protected areas.

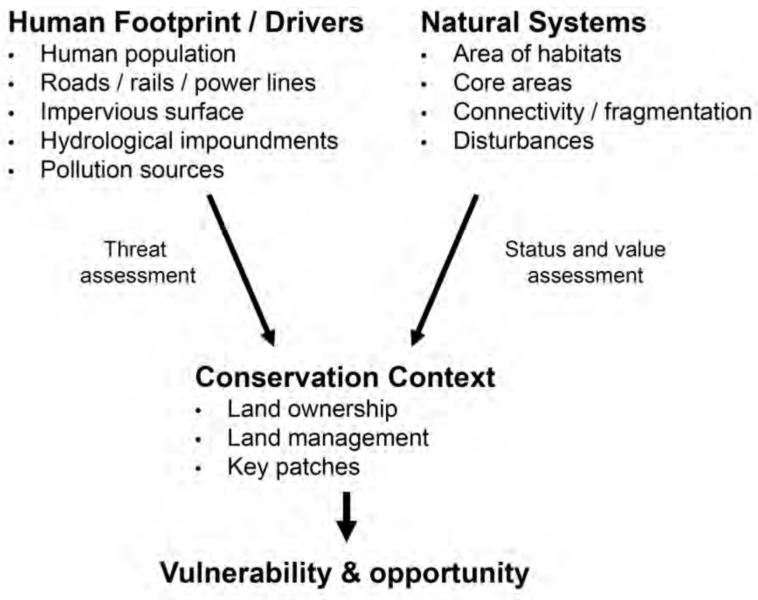

Fig. 1. Conceptual model used as a basis for landscape-scale assessment of parks.

Broad-scale data generally available include the human drivers represented in the model, and all of these drivers are well known to influence biodiversity and other park resources. Natural systems can be characterized in many ways, and the types of attributes in Figure 1 are a subset of important attributes that can be used to assess the context and condition of natural systems. The conservation context provides information that may be essential to decisions on land management. Svancara *et al*., (2009) conducted a national-level analysis by county of the conservation context of US national parks and refuges, and they discuss the use of this information to achieve conservation goals.

2.2 Landscape variables and data sources

Using the conceptual model as an overarching framework, we approached our analyses within the broader goals of the NPS landscape dynamics monitoring project, NPScape (http://science.nature.nps.gov/im/monitor/npscape). NPScape is designed to support landscape-scale monitoring conducted by the NPS Inventory and Monitoring Program (Fancy *et al*., 2009). Key NPScape objectives are to provide: a coherent conceptual and analytical framework for conducting landscape-scale analyses and evaluations that can inform decisions; Geographic Information System (GIS) data and maps at broad spatial scales that transcend the bounds of park-level data; well-documented methods founded on strong science that are readily repeatable and extensible with local data; and, assistance to parks in interpreting results.

In support of these objectives, NPScape produces and delivers a suite of landscape-scale datasets, methods, GIS scripts and tools, maps, and guidance reports to approximately 300 natural resource parks in the NPS system. Results from NPScape are intended to inform resource management and planning at multiple scales. Because the overarching goal of NPScape is to deliver information to parks across the entire NPS system, inputs are limited to data sources that cover broad spatial extents (i.e., regional to national).

NPScape incorporates a large number of datasets that can be roughly categorized into 'base layers' and 'variables'. Base layers are such things as topography, jurisdictional boundaries, hydrography, and the other geospatial data that are relatively static and that generally serve as covariates or provide a geopolitical context. The NPScape variables considered here address two major elements in our conceptual model: stressors and conservation context.

Important stressors are evaluated by measures of population, housing, roads, and impervious surface (a category of land cover). Conservation context was evaluated from the percentage of land in a protected status, and potential management partnerships from the number of different agency or institutional owners of conservation lands. The NPScape data sources and variables used in our analyses are described in Tables 1 and 2. Although NPScape includes a variety of other metrics related to natural land cover and landscape pattern, we did not use these in our national-level assessment because they require a more in-depth analysis at ecoregional scales. Our present focus on human drivers and conservation context is designed to serve as a foundation to these future studies.

Measure	**Source data**	**Years**	**Spatial resolution**	**Reference**
Population	US Census Bureau	2000	Census block groups	US Census Bureau, 2001; NPS, 2010a
Housing	Spatially Explicit Regional Growth Model (SERGoM)	2010	100 m cells	Theobald, 2005; NPS, 2010b
Roads	Environmental Systems Research Institute (ESRI)	Varies, up to 2005	Varies	ESRI, 2010; NPS, 2010c
Land cover	National Land Cover Data (NLCD)	2006	30 m cells	NPS, 2010d; Fry *et al.*, 2011
Conservation status	Protected Areas Database of the US (PAD-US)	Varies, up to 2010	Varies	NPS, 2011a; USGS Gap Analysis Program, 2011

Table 1. NPScape data sources used to compute the landscape variables (listed in Table 2).

Measure	**Variable**	**Units**	**Comments**
Population	Population density	count/km^2	Based on population totals
Housing	Housing density	# units/km^2	Based on mid-points of rural, exurban, suburban , and urban
	Rural housing	% area	0-6 units/km^2
	Exurban housing	% area	7-145 units/km^2
	Suburban housing	% area	146-1,234 units/km^2
	Urban housing	% area	>1,234 units/km^2
	Commercial/industrial	% area	Business – non-residential
Roads	Weighted road density	km/km^2	Highway weighted by a factor of 3, interstates by 10
Land cover	Impervious surface	%, area weighted	Anthropogenic sources only
	Developed	% area	Anderson Level I
	Developed open space	% area	Anderson Level II
	Low intensity development	% area	Anderson Level II
	Medium intensity development	% area	Anderson Level II
	High intensity development	% area	Anderson Level II
	Agriculture	% area	Anderson Level I
	Cultivated crops	% area	Anderson Level II
	Hay/pasture	% area	Anderson Level II
Conservation status	Protected	% area	Based on Gap Analysis Program (GAP) codes 1 and 2
	Landowner density	count/km^2	All owners of conservation lands, including NGO & private

Table 2. NPScape variables used in the present analyses.

In addition to NPScape variables, we included the percentage of each watershed in private ownership (derived from USGS Gap Analysis Program, 2011), and we obtained data on river impoundments and nitrogen (N) deposition. The US National Inventory of Dams database now contains more than 83,000 records of significant impoundments in US states and territories (US Army Corps of Engineers, 2010). These impoundments have dramatically altered hydrological flow regimes, sedimentation processes, inhibited or prevented biological migrations and other movements, and influenced virtually every ecological process in some catchments (Ward & Stanford, 1979). Following Sabo *et al.*, (2010) and Lawrence *et al.*, (2011) we used the density of impoundments (count/km^2) in the contributing watershed as an indicator of river fragmentation. A future refinement to our analyses is to include additional information on the characteristics of dams and their effects on aquatic resources. For example, watersheds in the eastern US tend to have a higher density of dams than western watersheds, but because the size of dams varies, the storage capacity as a portion of annual flow is nearly the same in the east and west (Sabo *et al.*, 2010). Thus western rivers generally have fewer but larger dams, so fragmentation is greater in the east but dams in the west alter hydrological dynamics more.

Anthropogenic activities now contribute more nitrogen (N) to the global cycle than all natural sources combined (Vitousek, 1997). Nitrogen is, or was, a key limiting element in many aquatic systems, but these limits are now exceeded in many parks (Baron *et al.*, 2011). Baron *et al.*, (2011) reviewed studies on N limitations in North American lakes. These studies revealed a consistent pattern of historical N limitations, especially in nutrient-poor environments typical of high elevations, and undisturbed temperate and boreal forests. Broad patterns of response to atmospheric N deposition further supported assertions that the majority of lakes in the Northern Hemisphere may have been N-limited prior to increased N deposition from anthropogenic sources. Atmospheric deposition of N has sufficiently altered the balance of N and Phosphorous (P) so that P limitations are now more commonly observed in North American lakes. These results emphasize the need to incorporate aspects of global change in broad-scale studies. Our analyses of N deposition are based on measurement of inorganic N wet deposition from the National Atmospheric Deposition Program (NADP), corrected for topographic precipitation differences using PRISM (Parameter-elevation Regressions on Independent Slopes Model) climate data as described in Baron *et al.*, (2011). These data underestimate total N deposition because they do not account for dry deposition. We did not attempt to account for terrestrial runoff or other N inputs.

2.3 Upstream watersheds and the headwater index

Watersheds that are either upstream or downstream with respect to a particular management unit may be calculated from Digital Elevation Models (DEMs) using GIS (Djokic & Ye, 1999). The park upstream watersheds considered here were calculated by NPScape using this basic methodology (NPS, 2011b). However, rather than relying on source DEM data, NPScape was able to take advantage of published DEM-derived datasets that serve as standardized pre-processed inputs to watershed calculations: the National Hydrology Dataset Plus (NHD Plus, 2010) vector flowlines, NHD Plus flow accumulation rasters, and NHD Plus flow direction rasters. In addition to these inputs, our calculations required NPS current administrative vector boundaries (NPS, 2011c) to determine pour points from the flowlines.

We used these four datasets (flowlines, flow accumulation grids, flow direction grids, and park boundaries) as inputs to the NPScape ArcGIS watershed toolbox (NPS, 2011b) to generate upstream catchments for all 261 natural resource parks in the contiguous US. Most parks had multiple upstream catchments originating from different sets of pour points around park boundaries. We dissolved catchment boundaries by park to derive final park upstream watersheds. Importantly, watersheds were computed with respect to the entire network of parks, meaning that upstream catchments were delineated based on the most proximate park in the NPS system. This decision was made in order to evaluate the landscape factors that relate directly to each park. Furthermore, because many parks occur in major river systems, it helped ensure that upstream watersheds were small enough to be practical for park management considerations, yet still ecologically relevant when considered in a larger NPS context.

From these outputs we applied a series of quality-control filters to eliminate park upstream watersheds with obvious inaccuracies (NPS, 2011b). We eliminated park watersheds where there were obvious errors with the source NHD Plus data, parks that were too small in relation to the spatial resolution and mapping accuracy of the source data, and parks that were in areas with complex hydrography (e.g., coastal, marine). These filters eliminated 110 of the 261 natural resource parks in the contiguous US, leaving a total of 151 focal parks and their contributing upstream watersheds that were considered in the analyses (Fig. 2).

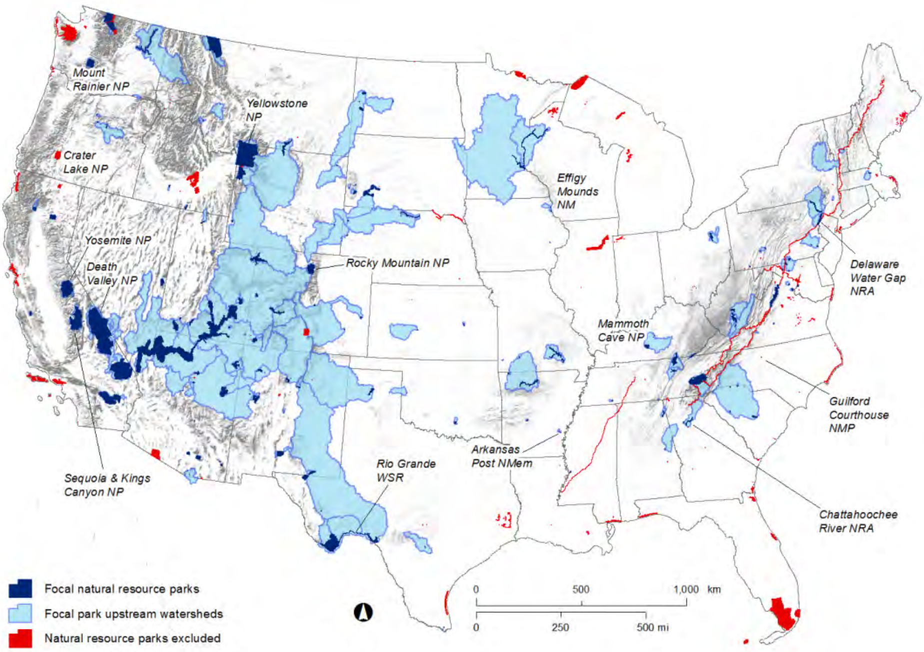

Fig. 2. Focal parks and upstream watersheds considered in the present analyses. Labelled parks are referred to in the text. NP = National Park, NRA = National Recreation Area, NMP = National Military Park, NM = National Monument, NMem = National Memorial, WSR = Wild and Scenic River.

We used the park boundaries and upstream watersheds to compute a geometric index of the degree to which a park includes its own headwaters. The headwater index was calculated by intersecting each park with its upstream watershed, then dividing that area by the total area of the upstream watershed. The resulting proportion ranged from zero (i.e., all upstream areas flowing into the park) to one (i.e., all upstream areas flowing out of the park).

2.4 Water quality variables and data sources

We derived estimates of water quality inside focal parks from two sources: the NPS Hydrographic and Impairment Statistics (HIS) database (http://nature.nps.gov/water/his/) and the Environmental Protection Agency (EPA) Storage and Retrieval System (STORET; EPA, 2011). The HIS provided data for each park on the total length of waterway (rivers, streams, canals, etc), as well as the total length of 'impaired' waterway identified by states according to the federal Clean Water Act sections 303(d) and 305(b). We used these two variables to estimate the percentage of total waterway in each focal park that was impaired (impairment data were not available for the Rio Grande Wild and Scenic River). In addition, we downloaded water chemistry data from STORET for the area within a 3 km buffer outside park boundaries for all parks in this study, restricting the data to observations from 1995-present, and to samples from rivers/streams, lakes, and reservoirs. Although STORET provides access to a very large number of chemical and biological variables, we restricted our analyses to acid neutralizing capacity, ammonia-nitrogen as N, dissolved oxygen, nitrogen-nitrate, pH, phosphate-phosphorus as P, dissolved solids, and specific conductance.

2.5 Analyses

We used a combination of univariate and multivariate methods to address our starting questions. Where possible we tried to emphasize univariate approaches, which are methodologically more intuitive and straightforward to interpret in a management context. However, because we considered a large number of landscape variables, we also needed a means to simplify analysis of the many correlated variables. To do so, we used principal component analysis (PCA) to identify broad orthogonal groupings of variables that explained most variation in park upstream watershed context. All statistical analyses were performed in R (R Development Core Team, 2011). Corresponding maps of select results were produced in ArcMap (ESRI, 2011); all maps are Albers equal area conic, NAD83.

The PCA was conducted using the landscape variables in Table 2, plus mean N deposition and dam density. Owing to non-normal distributions of the raw variables, arcsine transformations were first applied to all percentage (proportion) variables, and log transformations were used on all density variables. We excluded the headwater index from the PCA because we wanted to evaluate the major factors responsible for landscape-level change and management response (i.e., human drivers and conservation context) in park upstream watersheds, irrespective of their relatively static spatial geometries. We used the loadings of each transformed variable on principal components 1 and 2 (PC1, PC2) to interpret the meaning of each axis. Park-specific scores on PC1 and PC2 were then evaluated both geographically and in relation to the headwater index. For the latter, we regressed each principal component (dependent variable) on the arcsine transformed headwater index in order to explore the residuals and characterize the management potentials of non-headwater parks.

For water quality, we used Pearson's product moment correlation to characterize the association between the percentage of park waterway impaired (arcsine transformed) and PC1. We limited this correlation to PC1 because it explained the majority of landscape variation among park upstream watersheds. Meanwhile, an initial examination of STORET water quality data revealed implausible observations (outliers) for some variables. To reduce the effect of outliers in our analyses, we calculated the 95th percentile of the distribution for each variable and then multiplied this value by 3 (P3) and by 20 (P20). We removed all observations with values greater than P20. For observations with values between P3 and P20, we changed the observed value to the value of P3 (i.e., we truncated the distribution to ± P3). To obtain a single value for each variable and each park, we first calculated the median value of the observation for each site within a specific park area of analysis. We then calculated the mean of the site-specific medians for that area. We used linear regression with the park-specific mean values and our predictor variables (i.e., PC1, PC2, and a subset of NPScape variables in Table 2) to explore relationships between water quality and landscape attributes. After filtering the STORET data for date, location, and outliers, our analyses were based on usable data for 29-117 parks (mean = 78).

3. Results and discussion

3.1 What is the general context of park upstream watersheds?

Park upstream watersheds are potentially threatened by a number of landscape-level factors related to park-watershed geometry, housing development, habitat conversion and resource extraction, and N deposition (Fig. 3). Of the 151 park upstream watersheds considered, 81% have more than 50% of their watersheds extending beyond park boundaries, 77% have less than 50% area formally protected, 61% have greater than 10% rural development, and 37% have values for N deposition exceeding 3.5 kg N ha^{-1} yr^{-1} – a conservative critical load for most parts of the contiguous US (Baron *et al.*, 2011). Taken in combination, these numbers suggest that most parks do not directly control most of their watersheds, and that both physical and chemical stressors originating beyond park boundaries will likely affect water resources inside park boundaries. However, despite these challenges, it is equally noteworthy that park upstream watersheds are relatively unthreatened by converted land cover, including high-intensity human land use (Fig. 3). Of the 151 park upstream watersheds, just 12% are greater than 50% converted land cover, 17% are greater than 10% developed land cover, and 30% are greater than 10% agricultural land cover.

Several of these patterns merit further discussion. The low-level of protection afforded most park upstream watersheds is due in large part to the working definition of 'protected'. We consider parks and other areas 'protected' if they have permanent protection from conversion of natural land cover and a mandated management plan to maintain a primarily natural state. This definition follows from the US Geological Survey (USGS) Gap Analysis Program (GAP), which uses a series of four codes to rank areas based on their level of protection (USGS Gap Analysis Program, 2011). Our definition is based on GAP status codes 1 and 2 and includes most parks and all wilderness areas, but it excludes most lands managed by the Bureau of Land Management (BLM) and US Forest Service (USFS). These two Federal agencies combined manage approximately 1.8 million km^2, which irrespective of their use and reduced level of protection represent significant areas for natural resource conservation. We revisit this subject below in the context of watershed management opportunities for parks (Section 3.4).

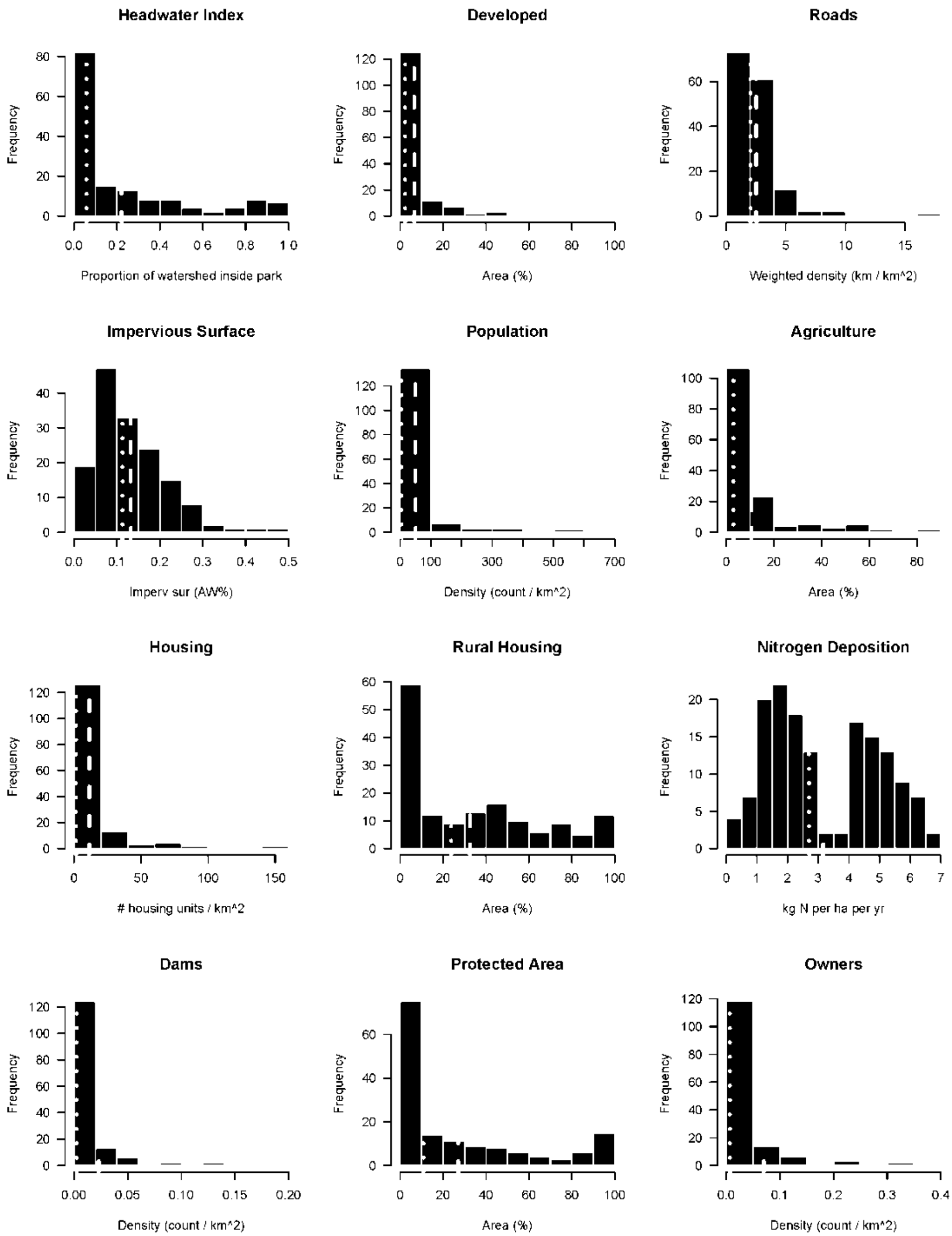

Fig. 3. Univariate distributions of select landscape variables for upstream contributing watersheds of 151 National Parks in the contiguous US. Dashed lines show means; dotted lines show medians.

Park upstream watersheds are bimodally distributed with respect to N deposition (Fig. 3). This bimodality is strongly influenced by a combination of longitude and elevation. Based on critical loads from Baron *et al*., (2011), all park upstream watersheds in the east exceed the critical load of 3.5 kg N ha^{-1} yr^{-1} reported for the northeast; Yosemite, Sequoia, and Kings Canyon National Parks all exceed the critical load of 1.5 kg N ha^{-1} yr^{-1} reported for the Sierra Nevada; and, all parks in the Central Rockies exceed the critical load of 1.0 kg N ha^{-1} yr^{-1} reported for the Rocky Mountains (Fig. 4). Hence, despite geographic variation in N deposition across parks in the contiguous US, most park upstream watersheds considered here have values exceeding critical loads for their respective geographies. Future work is needed to incorporate more detailed geographic estimates of critical loads for N deposition.

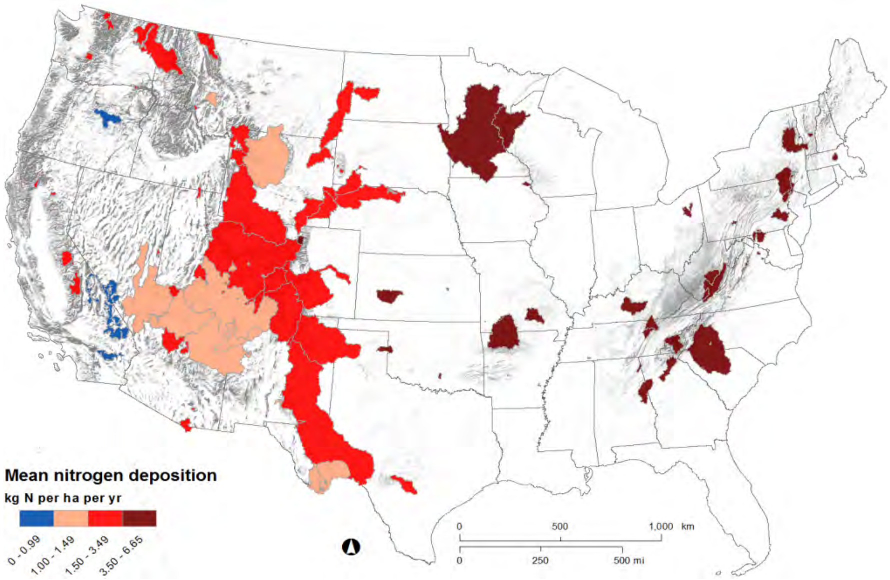

Fig. 4. N deposition in park upstream watersheds, with legend categories reflecting critical loads described by Baron *et al*., (2011) for different areas of the US.

Rural development (<7 housing units km^{-2}; Theobald, 2005) has already occurred over extensive areas in most park upstream watersheds, and there is great concern about the rate of development of rural landscapes around parks (Hansen *et al*., 2005; Wade & Theobald, 2009; Radeloff *et al*., 2010). Increases in the extent of low-density housing in previously undeveloped areas has numerous biological impacts (Hansen *et al*., 2002, 2005) and housing development is increasingly recognized as a primary driver of ecological processes and as a threat to biodiversity (McKinney, 2002; Miller & Hobbs, 2002). In the Greater Yellowstone Ecosystem, which is threatened by exurban development, riparian habitat, elk winter range, migration corridors, and other important habitat and biodiversity indices are expected to experience substantial conversion (between 5% and 40%) by 2020 (Gude *et al*., 2007). Hence, this driver will be increasingly important for ongoing and future management of park watersheds.

Lastly, dam density is characteristically low in most park upstream watersheds (Fig. 3; mean = 0.02 dams/km^2), but it is important to note that ecologically relevant thresholds for this variable are also low and likely close to this mean for many natural resources, especially when considered in the context of dam size. For example, a single large dam may affect water temperatures and benthic communities for hundreds of kilometres downstream (Baxter, 1977). In addition, higher densities of small dams may have cumulative effects on physicochemistry and macroinvertebrate diversity that exceed those of large dams (Mantel *et al.*, 2010). Single dams may also create serious obstacles to the long-range movement of fish, either upstream (e.g., anadromous salmon) or downstream (e.g., catadromous eels). In brief, dams have pervasive and varied effects on aquatic resources (Ward & Stanford, 1979), and the analyses presented here would greatly benefit from an expanded set of ecologically informative variables and thresholds related to impoundments.

3.2 Which landscape factors explain most variation in park upstream watersheds?

Using the human driver and conservation context variables shown in Figure 3, plus additional physical landscape variables described under landscape variables and data sources (Section 2.2), we conducted a PCA to understand which of the 21 landscape factors explained most of the among-park variation in upstream watershed context. Principal components 1 and 2 (PC1, PC2) explained 51% and 15% (respectively) of the variation (66% total). PC1 loaded positively on several variables related to urban development, while PC2 loaded positively on variables related to both agriculture and N deposition, and negatively on the amount of protected area (Table 3). According to both axes, higher values (denoting higher urban development, agriculture, and N deposition; less protected area) are associated with parks east of the Rocky Mountains (Fig. 5). Dam density loaded most strongly on PC4, but this axis explained only 6% of the variation and is thus not shown.

Variable	PC1	PC2
Urban development	0.29	-0.12
Low intensity development	0.29	-0.12
Population density	0.28	0.04
Suburban housing	0.28	-0.14
Urban housing	0.27	-0.12
Medium intensity development	0.27	-0.18
Developed open space	0.27	-0.01
High intensity development	0.26	-0.18
Agriculture	0.17	0.42
Cultivated crops	0.09	0.38
Rural housing	-0.04	0.38
Pasture/hay	0.18	0.31
Nitrogen deposition	0.16	0.30
Protected area	-0.14	-0.28
Commercial/industrial	0.25	-0.21
Housing density	0.24	0.16
Exurban housing	0.23	0.20
Impervious surface	0.22	-0.08
Weighted road density	0.17	-0.08
Owner density	0.06	-0.13
Dam density	0.13	-0.04

Table 3. Principal component analysis loadings by variable for axes 1 and 2 (PC1, PC2). Values are grouped on each column to facilitate interpretation of the axes.

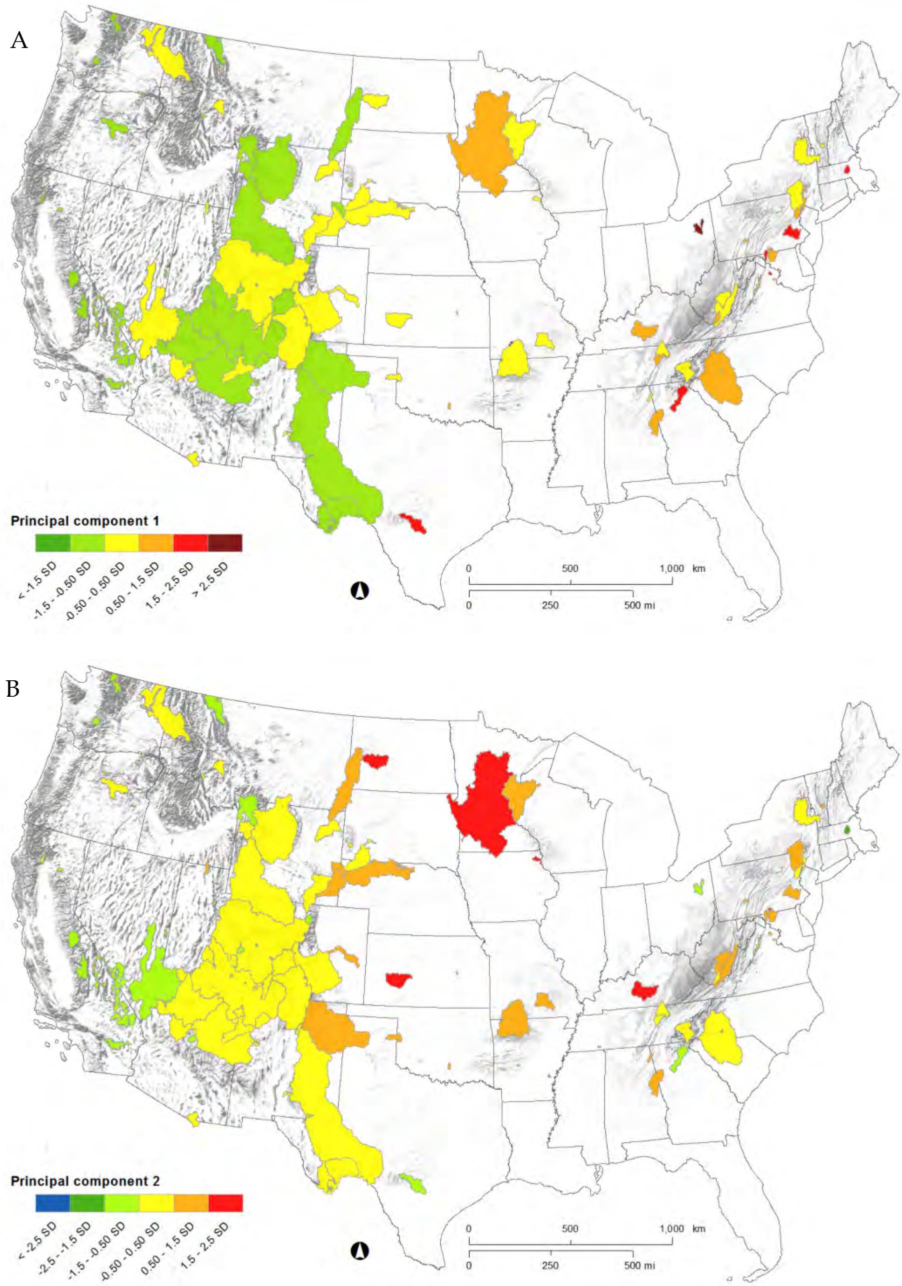

Fig. 5. Principal component scores 1 (A) and 2 (B) shown for park upstream watersheds. Orange and red colours denote watersheds that have higher PC scores (higher threat), in units of standard deviations (SD).

3.3 What can we infer about the condition of park freshwater resources?

Based on the percentage of impaired waterway, 62%, 64%, and 70% of parks (respectively) have less than 5%, 10%, and 20% of their total waterways in non-compliance with federal Clean Water Act sections 303(d) and 305(b) (Fig. 6). However, in this context it is important to note that 'impairment' standards vary by state and are generally less stringent than critical ecological thresholds in most parks. Furthermore, the sources of waterway 'impairment' do not all originate from park upstream watersheds. For these reasons, one would not *a priori* expect a substantial amount of among-park variation in waterway impairment to be explained solely by the landscape dynamics of upstream watersheds. We find that the percentage of park waterway impaired is positively correlated with PC1, and that this variable explains approximately 26% of variation.

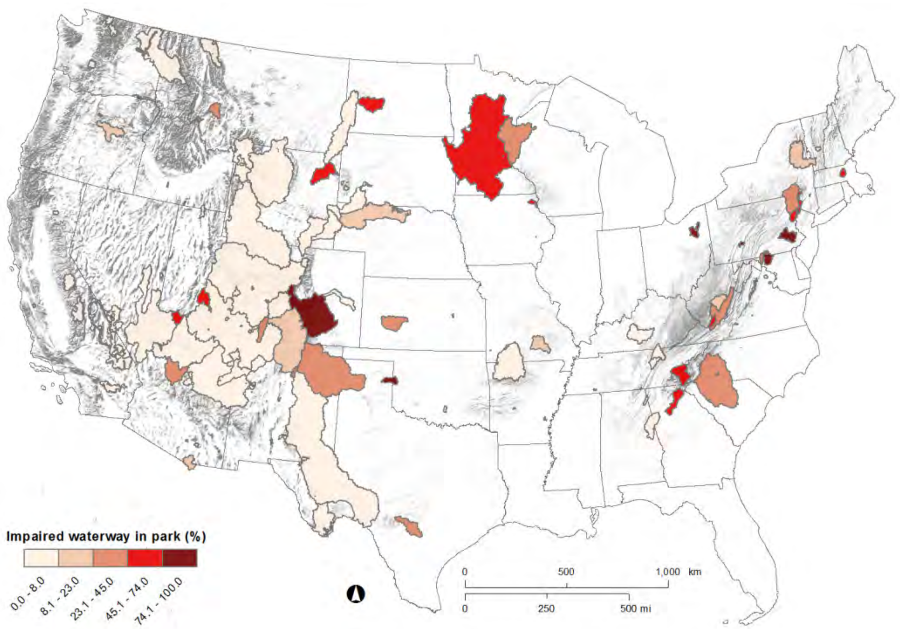

Fig. 6. The percentage of impaired waterway in focal parks. Note that the summary statistic was calculated for each park, but results are symbolized here by park upstream watershed to facilitate comparisons with the other maps.

When compared to ecological threshold values for poor or good water quality (e.g., Van Sickle *et al.*, 2006; Wazniak *et al.*, 2007; Riva-Murray *et al.*, 2010), water quality in and near most parks is in a good range. These results reflect the landscape location of most park watersheds, which tend to include a high portion of natural land cover and a relatively small area of cropland or intensive development. Using simple regression analyses, we generally found weak relationships between STORET water chemistry variables and watershed landscape variables. Certain attributes of the data likely contributed to our inability to link these factors. We wished to evaluate the ability of large, broad-extent databases to inform regional and national-scale analyses, and we thus began our analyses with approximately 2 million observations. The large number of observations required automated processes to screen data. For these preliminary analyses, we did not attempt to correct for factors such as season, variation in sampling effort, or flow regime. Despite the absence of strong statistical associations between water chemistry and landscape context, regional patterns were apparent for most of the chemistry variables we examined, such as phosphate-phosphorus (Fig. 7).

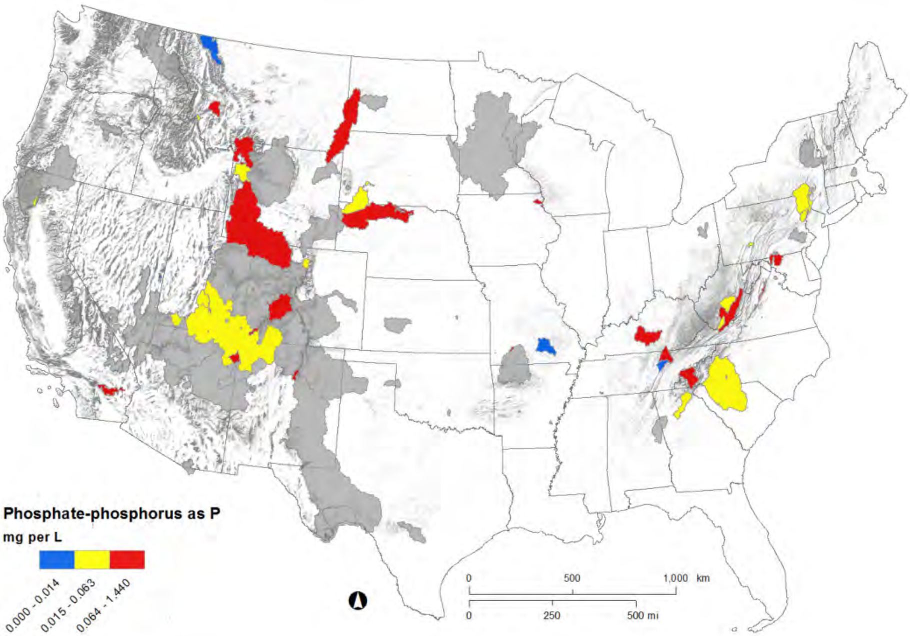

Fig. 7. Concentration of phosphate-phosphorus in focal parks, with legend categories reflecting conservative critical thresholds described by Van Sickle *et al.*, (2006) for total P. Note that concentrations were calculated for each park with 3 km buffer, but results are symbolized here by park upstream watershed in order to facilitate comparisons with the other maps. Watersheds in light grey denote parks that were not sampled for this metric.

The absence of strong statistical relationships between landscape and water quality variables in our national-scale assessment indicates the need for more sophisticated analyses when working at these very broad scales and with generalized databases. Other studies have found considerably stronger relationships between land cover and water chemistry (e.g., King *et al.*, 2005; Wickham *et al.*, 2005; Riva-Murray *et al.*, 2010). Our future efforts will include more sophisticated processes for screening water chemistry data, and additional analyses. For example, King *et al.*, (2005) evaluated a water quality index based on binary (0, 1) values for predictor variables that were above or below a quality threshold. Their index was the sum or four predictor variables. The binary transformation effectively addresses issues with high variance in the predictor variables, and it simplifies estimation and interpretation of the index. In addition, evaluations of the relative contributions of land cover versus broad-scale environmental setting to determining water chemistry are ambiguous and clearly scale-dependent (e.g., compare King *et al.*, 2005; Wickham *et al.*, 2005; Goldstein *et al.*, 2007). Our analyses combined all samples for a given park so we could reach conclusions at the scale of an aggregated park upstream watershed. In some cases, this procedure merged samples from contributing watersheds that differed dramatically in land use patterns and threats to small-scale watersheds (e.g., Delaware Water Gap; Gross *et al.*, 2011). Future analyses of land use effects on park water resources will likely need to resolve data at a finer spatial scale, perhaps in the form of hierarchical models. An appropriate management-relevant improvement would be to conduct local and regional-scale analyses on watersheds upstream of sampling sites, and then extrapolate these results to park watersheds within relevant ecological regions (Rohm *et al.*, 2002).

3.4 What are the major challenges and opportunities for managing park upstream watersheds?

Among the 151 focal parks, the big challenges identified by these analyses for managing park upstream watersheds relate to three major categories: urban development (PC1, Fig. 5A), agriculture and diffuse rural development (PC2, Fig. 5B), and N deposition (Fig. 4). Habitat fragmentation and alteration due to dams is undoubtedly a fourth major challenge, but one that we were unable to adequately quantify in this analysis. Nonetheless, the assessment of landscape context revealed that practically all parks are threatened in their respective geographies by N deposition, and parks east of the Rocky Mountains are especially threatened by development and agriculture. Importantly, this is not to say that parks in the western US are unthreatened by historical changes in land cover and land use. When compared to eastern parks the upstream watersheds of western parks are not presently as impacted by these factors, but critical ecological thresholds may still be exceeded in certain areas (e.g., Porter *et al.*, 2005; Porter & Johnson, 2007). Furthermore, human population and housing projections suggest that many western parks will be increasingly challenged by development pressures in the coming decades (Theobald, 2005; Radeloff *et al.*, 2010).

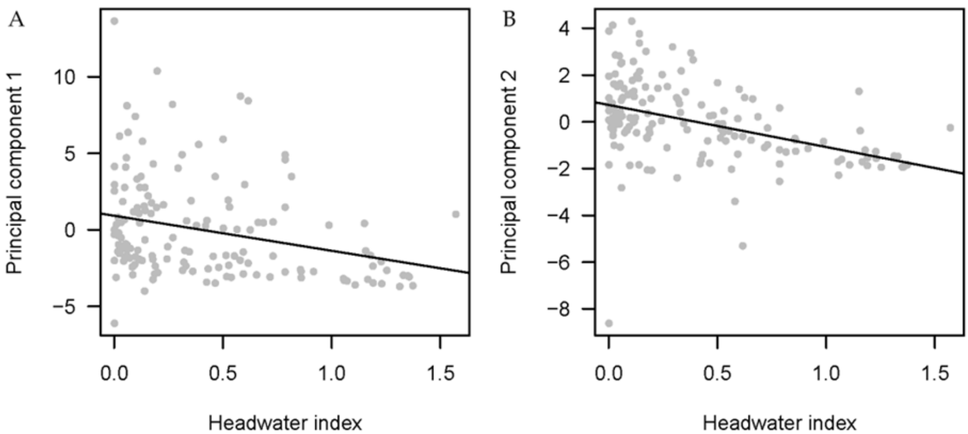

Fig. 8. Principal component scores 1 (A) and 2 (B), versus the headwater index (arcsine transformed), for 151 focal park upstream watersheds.

In the face of these monumental challenges, opportunities for managing park upstream watersheds are generally positively related to the headwater index – the proportion of the upstream watershed that exists within park boundaries. Managers of headwater parks obviously have the greatest direct management control over upstream watershed issues. Examples in this category include large National Parks in the western US: Yellowstone, Rocky Mountain, Sequoia and Kings Canyon, Yosemite, and Mount Rainier. In addition, owing to the strong human land cover and land use variables loading into PC1 and PC2, the two axes are negatively related to the headwater index (Fig. 8). From these relationships we can identify particular parks that – based on their headwater index – have characteristically low values for PC1 or PC2. In effect, these are parks with upstream watersheds that are relatively unchallenged by human drivers of landscape change, at least considering that significant portions of their upstream watersheds lie beyond park boundaries. Management opportunities for these parks lie in working collaboratively with other land owners to maintain protection of the upstream watershed. Example parks in this regard include Guilford Courthouse National Military Park (PC1), Chattahoochee River National Recreation Area (PC1), Effigy Mounds National Monument (PC2), and Arkansas Post National Memorial (PC2).

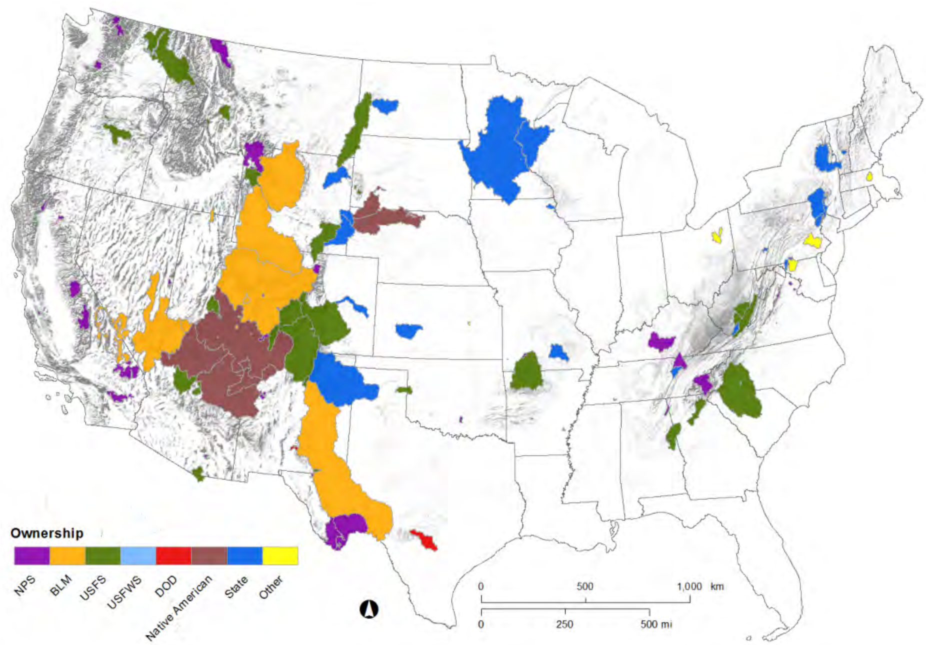

Fig. 9. Dominant owners of conservation land in focal park upstream watersheds. NPS = National Park Service, BLM = Bureau of Land Management, USFS = US Forest Service, USFWS = US Fish & Wildlife Service, DOD = Department of Defense.

Conservation partnerships are challenging to promote, in part due to varied and sometimes conflicting missions of the partners, and perhaps also due to an insular history of managing for resources within ownership boundaries. Nevertheless, partnership opportunities may initially be evaluated using a simple landscape metric like the density of landowners that manage lands for conservation. Although this variable did not emerge as a major factor explaining among-park variation in watershed context (Table 3), it can be very useful for particular parks seeking to understand the potential diversity of partners that need to be engaged, as well as the dominant landowners that control most areas upstream (Fig. 9). The recognition that neighbouring landowners share a common responsibility for managing resources in the face of landscape-level anthropogenic change has motivated actions at local to national scales to form new partnerships. It has also recently stimulated the establishment of the Department of Interior (DOI) Landscape Conservation Cooperatives (LCCs) and regional Climate Science Centers (DOI Secretarial Order 3289).

Private lands not held for conservation pose a separate and distinct set of challenges and opportunities for managing park upstream watersheds. Although not shown in Figure 9, non-conservation private lands encompass approximately 61% of the US, and they thus dominate many park upstream watersheds (Fig. 10). While it is challenging to coordinate a large number of different private landowners, such coordination may be facilitated when private lands share a common land use. For example, private landowners in an area dominated by cultivated crops may share problems with ditch erosion (i.e., increased time and costs with ditch maintenance), which also poses sedimentation challenges to water resources in a downstream park. Despite different concerns over the threat, there would be a united interest in identifying creative solutions to the problem. Non-governmental organizations (NGOs) have traditionally played an especially important role in coordinating private-public partnerships. Such partnerships may also be promoted through newly established LCCs.

3.5 Next steps

There are several important ways to build upon the analyses presented here. To evaluate a wider range of anthropogenic landscape stressors and pollutants, it is important to consider other areas of analysis besides upstream watersheds. Other ecologically informative areas of analysis could include downstream watersheds, ecoregions, or a local area that is proximate to the management unit (e.g., 30 km buffer or a protected area centered ecosystem, PACE; Hansen *et al.*, 2011). Using these varied areas of analysis would extend our framework to consider other water quality response variables that are affected by pollutants that traverse the landscape in different ways. In addition to new water resource response variables, it would also be useful to extend the analyses to consider point source drivers and their spatial proximity to parks based on flow length. However, given that these can vary so dramatically by geography, both in terms of point source type and magnitude, the explanatory power of these additional landscape predictors may prove most useful in analyses of parks at local to regional scales. Beyond the human drivers and conservation context of park upstream watersheds considered here, future analyses need to explicitly consider the ecological benefits and buffering potentials of natural systems. While some of these variables – like the percentage of natural land cover types – will be inversely correlated with many of the landscape stressors (e.g., percentage urban, percentage agriculture), others related to landscape pattern (Riitters *et al.*, 1995, 2007, 2009a, 2009b) and habitat connectivity (Hilty *et al.*, 2006; Theobald, 2006; Goetz *et al.*, 2009; Galpern *et al.*, 2011) will provide key management insights at local to regional scales.

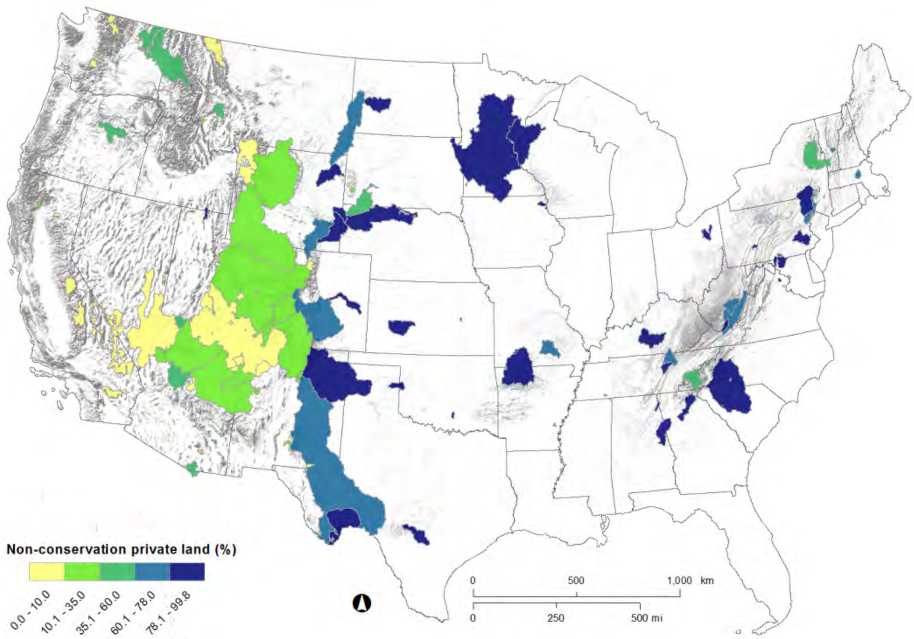

Fig. 10. Percentage of non-conservation private land in focal park upstream watersheds.

4. Conclusion

We demonstrate a GIS framework for quantifying broad-scale landscape dynamics of park upstream watersheds and interpreting those analyses in the context of park water resources and watershed management potential. The framework is valuable for assessing NPS-wide opportunities and challenges associated with preserving water resources across an entire network of management units. Because it is founded on publically available data and methods, which were chosen based on mechanistic relationships between landscape-scale factors known to affect protected areas (Hansen & DeFries, 2007) and water resources (Allan, 2004), the framework may be readily applied to other systems of protected areas in the US, and also to protected areas in other parts of the world with comparable landscape and water resource data. When applied to 151 focal parks in the contiguous US, we demonstrate how (1) major anthropogenic stressors upstream from parks vary geographically, both in terms of magnitudes and critical ecological thresholds (e.g., N deposition), (2) water chemistry and impairment observations from most parks are within a good range, reflecting the overall landscape context of parks, (3) certain non-headwater parks are surprisingly unchallenged by upstream stressors that affect water quality, and (4) parks vary dramatically in terms of the public and private partnership opportunities for coordinating watershed management. While these findings do not provide park-specific recommendations for managing water resources, they are foundational to helping us better understand park watersheds and water quality in a comparative NPS-wide context, which in turn may inform interpretations of site-level analyses at policy-relevant scales.

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