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Exposure of U.S. National Parks to land use and climate change 1900–2100

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Abstract. Many protected areas may not be adequately safeguarding biodiversity from human activities on surrounding lands and global change. The magnitude of such change agents and the sensitivity of ecosystems to these agents vary among protected areas. Thus, there is a need to assess vulnerability across networks of protected areas to determine those most at risk and to lay the basis for developing effective adaptation strategies. We conducted an assessment of exposure of U.S. National Parks to climate and land use change and consequences for vegetation communities. We first defined park protected-area centered ecosystems (PACEs) based on ecological principles. We then drew on existing land use, invasive species, climate, and biome data sets and models to quantify exposure of PACEs from 1900 through 2100. Most PACEs experienced substantial change over the 20th century (>740% average increase in housing density since 1940, 13% of vascular plants are presently nonnative, temperature increase of 1°C/100 yr since 1895 in 80% of PACEs), and projections suggest that many of these trends will continue at similar or increasingly greater rates (255% increase in housing density by 2100, temperature increase of 2.5°–4.5°C/100 yr, 30% of PACE areas may lose their current biomes by 2030). In the coming century, housing densities are projected to increase in PACEs at about 82% of the rate of since 1940. The rate of climate warming in the coming century is projected to be 2.5–5.8 times higher than that measured in the past century. Underlying these averages, exposure of individual park PACEs to change agents differ in important ways. For example, parks such as Great Smoky Mountains exhibit high land use and low climate exposure, others such as Great Sand Dunes exhibit low land use and high climate exposure, and a few such as Point Reyes exhibit high exposure on both axes. The cumulative and synergistic effects of such changes in land use, invasives, and climate are expected to dramatically impact ecosystem function and biodiversity in national parks. These results are foundational to developing effective adaptation strategies and suggest policies to better safeguard parks under broad-scale environmental change.

Key words: climate change; policy; U.S. National Parks; vulnerability assessment.

Introduction

Protected areas (PAs) are defined as "areas of land and/or sea especially dedicated to the protection and maintenance of biological diversity..." (Dudley and Stolton 2008). As such, PAs are cornerstones of the global strategy for safeguarding nature (Possingham et al. 2006:510). The rationale for the PA approach is that restricting human activities within protected areas will allow natural processes and native species to persist (Gaston et al. 2008). This approach recognizes that some native species will likely be reduced on the lands needed by humans for food, shelter, and other resources, and thus PAs are set aside as critical strongholds for such species and their habitats. Since the PA concept began to be widely applied over the last century, however, evidence has increased that human activities are altering the biosphere in the form of climate change, land use intensification, pollution, spread of invasive species, and other factors (IPCC 2007). Recognition of human-induced global change raises questions about the viability of the core concept of PAs as areas relatively free of human influence (Caro et al. 2012).

Human impacts on PAs can be conceptualized as global and regional (Fig. 1). Human-induced global change may be manifest within PAs as changes in climate or pollution that directly influence ecosystem processes and organisms in the PAs. Similarly, human transport of species outside their native ranges can result in noxious nonnative species establishing in PAs and displacing native species. The possible influence on PAs

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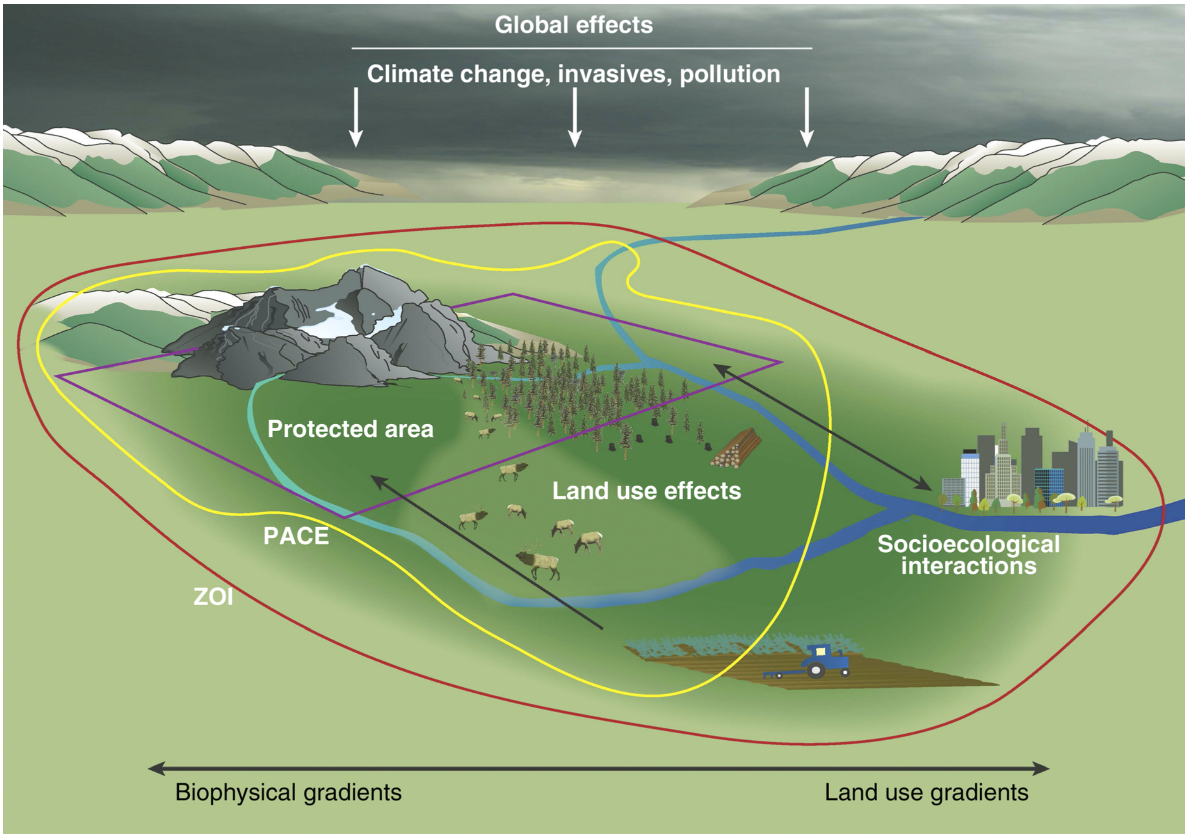

FIG. 1. Conceptual model of a protected area (PA) depicted in the context of regional and global human influences. The PA lies within a protected-area centered ecosystem (PACE) (see *Introduction*). The PACE is mapped based on ecological flows, crucial habitats, effective habitat size, and human edge effects. The still larger zone of interaction (ZOI) includes the region of strong two-way interactions between the PACE and surrounding human communities, involving ecosystem services, economics, policy, and social values (DeFries et al. 2010). Both the PA and the human system are influenced by biophysical gradients and land use gradients. Humans also influence PAs through alteration of global climate and other factors. PA managers are challenged to consider both regional and global influences in order to develop effective management strategies.

of regional factors such as land use intensification in the surroundings is less obvious because it is occurring outside of PA boundaries. Ecologists have increasingly learned, however, that the properties of ecosystems and species populations are dependent on their spatial dimensions (Chapin et al. 2011) and reducing the area of an ecosystem can change these properties (Fahrig 2003). The boundaries of few PAs were designated to ensure ecological completeness (Newmark 1985). Thus they may exclude portions of the spatial domain of nutrient flows, organism movements, disturbance regimes, and population dynamics centered on the protected areas (Shafer 1999). Land use intensification on surrounding lands may disrupt these flows and alter ecological processes and biodiversity within PAs (Hansen and DeFries 2007). Accordingly, the term protected-area-centered ecosystem (PACE) has been used to describe areas wherein human activities may negatively influence ecological processes and the viability of native species within the PA (Hansen et al. 2011).

A critical limitation in our current knowledge of PAs is the rate and ecological consequence of global and regional change within individual PAs and across national networks of PAs. Studies to date have found that land use is intensifying around many protected areas (DeFries et al. 2005, Wittemyer et al. 2008, Radeloff et al. 2010, Wade and Theobald 2010, Davis and Hansen 2011, Leroux and Kerr 2013). Nonnative plants are known to be widespread within U.S. National Parks (Allen et al. 2009). Projections for climate change across the global network of PAs indicate that only 8% of PAs will maintain current average temperatures throughout the next century (Loarie et al. 2009). However, systematic assessments of past change and projected future change in stressors of PAs and ecological consequences are lacking in the United States and internationally. Consequently, knowledge is insufficient to direct research and management toward those PAs that are most vulnerable to human-induced change.

Organizations that oversee PAs have increasingly recognized the need to assess their vulnerability to global change. The World Commission on Protected Areas is developing ways to evaluate climate change impacts (Dudley et al. 2010). Within the United States, the Department of Interior (DOI) launched in 2009 new programs on climate science and management (U.S. Department of the Interior 2009). Consistent with these programs, the National Park Service (NPS) published a climate change response strategy (National Park Service 2010), which states that the NPS "will conduct scientific research and vulnerability assessments necessary to support NPS adaptation, mitigation, and communication efforts." In the face of global change, an NPS science advisory panel recommended in 2012 that, "The overarching goal of NPS resource management should be to steward NPS resources for *continuous change that is not yet fully understood, in order to preserve ecological integrity*..." (Colwell et al. 2012:11). Recognizing that U.S. National Parks have a history of being managed individually and within their boundaries, this panel emphasized that management should encompass a geographic scope beyond park boundaries, consider longer time horizons, and evaluate park units as elements of a national network.

A promising framework for climate adaptation planning was recently developed by an inter-organizational committee (Glick et al. 2011) and is consistent with the NPS climate change strategy (National Park Service 2010). The four steps of the framework are to (1) identify conservation targets, (2) assess vulnerability, (3) identify management options, and (4) implement management options. The vulnerability assessment in Step 2 identifies what is at risk and why. Vulnerability is evaluated in terms of three components. *Exposure* is the degree of change in climate and land use, which are key drivers of ecological processes and biodiversity. *Sensitivity* is the degree to which species and ecological processes respond to a given level of exposure, largely based on the environmental tolerances of organisms. Exposure and sensitivity determine the potential impact on the resource of interest. *Adaptive capacity* is the ability of a system to adjust to the elements of exposure. It is the interaction of potential impact and adaptive capacity that determines vulnerability.

The goal of this study is to systematically apply elements of the Glick et al. (2011) approach to a network of U.S. National Parks in order to identify which parks are undergoing the highest rates of exposure to individual and cumulative elements of human-induced change and to demonstrate methods by which vulnerability assessments can be operationalized across networks of PAs in the United States and globally. The objectives are as follows: (1) Quantify past exposure of PAs to land use, climate change, and invasive species during 1900–present. (2) Project potential future exposure to land use and climate change and potential impact of climate change to vegetation distributions for the period 2000–2100. (3) Evaluate how well historical exposure predicts potential future exposure and the magnitude of exposure during the full 1900–2100 period.

Methods

We focused on past exposure to land use, climate change, and invasive species across the PACEs in and around 57 U.S. National Park units. Exposure of the PACEs to land use, invasive species, and climate change over the last century was quantified by change in housing density during 1940 to 2010, change in climate during 1895 to 2009, and the current presence of nonnative plants. The ecological impacts of this past exposure is not reported because data are inadequate on response of species and ecosystems to these changes in and around U.S. National Parks. Projected exposure to 2100 was evaluated from projected changes in housing density and climate consistent with the main storylines of the IPCC Special Report on Emissions Scenarios (SRES; IPCC 2001). Potential impact of future climate change was summarized as the percentage of each PACE projected to undergo a shift in biome type based on climate. Cumulative exposure to land use, climate change, and invasive species was represented graphically to facilitate comparison among PACEs within the United States. The extent to which projected future exposure to housing density and temperature is correlated with past exposure is reported, as is change in exposure for the full 1900–2100 time period. We conclude by discussing the utility of our assessment for informing adaptation planning within PACEs and informing NPS policy.

For this study, we used the PACEs delineated around the larger U.S. National Park units (national parks, monuments, and recreation areas; referred to collectively as "parks") in the contiguous United States in our previous study (Davis and Hansen 2011; see also Table 1, Fig. 2). The selected parks represent a wide distribution of climate and land use gradients, and are primarily managed for natural values, biodiversity, and recreation. Some parks were combined for analysis because they shared borders or were managed as a single unit, leading to a total of 49 different analysis units. The PACEs were delineated based on three ecological criteria: contiguity of surrounding natural habitat, watershed boundaries, and extent of human edge effects (see Davis and Hansen 2011).

Land use, invasive species, and climate 1900–2010

We represented land use as change in housing density from 1940 to 2010, current land allocation, and percentage of private land in PACE currently in an "undeveloped" condition. Housing density is one important measure of land use intensity that is relevant to ecological impacts (Theobald 2005). Housing density block data for 2010 was obtained from the U.S. Census. Housing density (units/km^2, 270-m resolution) at a given decade in the past, D_t, was assumed to be proportional to the ratio of the number of housing units in a county

TABLE 1. Land use properties of the protected-area centered ecosystems (PACEs) surrounding the U.S. National Park units included in this study.

Park	PACE code	Date of park establishment	Park area (km^2)	PACE : park area ratio	Percentage of PACE: In public land	Percentage of PACE: Undeveloped private land in 2000	PACE typology
Arches	ARCH	1929	309	29	93	88.9	wildland protected
Badlands	BADL	1929	982	15	20	95.3	wildland developable
Big Bend	BIBE	1935	3 291	8	20	99.6	wildland developable
Bighorn Canyon	BICA	1964	484	33	33	96.1	wildland developable
Big South Fork	BISO	1974	496	32	27	56.0	exurban
Big Thicket	BITH	1974	359	15	5	63.4	wildland developable
Blue Ridge Parkway	BLRI	1936	366	81	15	19.6	exurban
Buffalo River	BUFF	1972	389	31	23	64.0	agriculture
Canyon de Chelly	CACH	1931	375	18	6	92.3	wildland developable
Colorado River†	CORI†	1908–1964	18 295	5	76	93.6	wildland protected
Crater Lake	CRLA	1902	736	7	85	92.9	wildland protected
Craters of the Moon	CRMO	1924	1 901	9	76	93.8	wildland protected
Death Valley	DEVA	1933	13 764	4	79	80.7	wildland protected
Delaware Water Gap	DEWA	1965	278	24	20	15.0	exurban
Dinosaur	DINO	1915	853	22	74	94.7	wildland protected
El Malpais	ELMA	1987	473	17	38	96.1	wildland developable
Everglades, Big Cypress	EVER	1934, 1974	9 179	3	62	57.0	urban
Glacier	GLAC	1910	4 080	5	80	77.9	wildland protected
Great Basin	GRBA	1922	312	21	93	95.7	wildland protected
Great Sand Dunes	GRSA	1932	496	18	43	91.1	wildland developable
Great Smoky Mountains	GRSM	1926	2 098	7	43	15.1	exurban
Guadalupe Mountains	GUMO	1966	356	21	44	99.5	wildland developable
Joshua Tree	JOTR	1936	3 211	7	74	62.9	wildland protected
Lake Roosevelt	LARO	1946	424	39	22	91.3	agriculture
Lassen Volcanic	LAVO	1907	434	9	47	92.2	wildland developable
Missouri River	MNRR	1978	279	71	6	82.4	agriculture
Mojave	MOJA	1994	6 433	3	94	88.9	wildland protected
Mount Rainier	MORA	1899	952	6	70	80.2	wildland protected
New River Gorge	NERI	1978	285	28	15	42.1	exurban
North Cascades Complex	NOCA	1968	2 756	6	91	73.8	wildland protected
Olympic	OLYM	1909	3 700	5	47	65.1	wildland developable
Organ Pipe Cactus	ORPI	1937	1 338	8	43	99.1	wildland developable
Ozark	OZAR	1964	333	29	29	82.8	wildland developable
Petrified Forest	PEFO	1906	903	11	25	97.3	wildland developable
Pictured Rocks	PIRO	1966	298	17	60	82.5	wildland developable
Point Reyes, Golden Gate	POGO	1962, 1972	617	10	28	32.6	urban
Redwood	REDW	1968	468	15	60	82.2	wildland developable
Rocky Mountain	ROMO	1915	1 080	8	83	53.6	wildland protected
Saint Croix	SACN	1968	396	21	15	66.5	agriculture
Saguaro	SAGU	1933	378	48	55	57.2	exurban
Santa Monica Mountains	SAMO	1978	619	9	33	27.6	urban
Shenandoah	SHEN	1926	782	14	16	19.8	exurban
Sleeping Bear Dunes	SLBE	1970	284	16	36	22.7	exurban
Theodore Roosevelt	THRO	1947	285	30	41	95.8	agriculture
Voyageurs	VOYA	1971	829	11	77	80.6	wildland protected
White Sands	WHSA	1933	617	17	34	74.3	wildland developable
Yellowstone Grand Teton	YELL	1872	10 159	3	93	73.3	wildland protected
Yosemite, Sequoia-Kings Canyon	YOSE	1890	6 521	3	90	66.2	wildland protected
Zion	ZION	1909	598	14	69	87.1	wildland protected
Mean			2 140	18	49	72.62	

Note: Date of establishment (authorization, proclamation, or initial recognition by originating agency) is from http://www.nps.gov/applications/budget2/documents/chronop.pdf. All other data are from Davis and Hansen (2011).

† Colorado River parks are Canyonlands, Capitol Reef, Glen Canyon, Grand Canyon, and Lake Mead.

C_t (or tract when time $t \geq 1980$) to the number of housing units in 2010, C_0, as follows: $D_t = D_0 \times (C_t/C_0)$. A 5-km moving window was used to smooth abrupt changes that can occur at the edges of counties or tracts. The historical county and tract data sets are from the Minnesota Population Center (2011). We additionally report three metrics from Davis and Hansen (2011), percentage of PACE in public lands, percentage of private land developed, and land use typology, in order to characterize land use change beyond housing density. The percentage of each PACE's area in public vs. private and private protected ownerships was derived from the Protected Area Database of the United States, v4.5 (Conservation Biology Institute 2006). This metric is of interest because unprotected private lands have the potential to be developed to more intense land uses such as agricultural, suburban, and urban areas while public lands and private lands in conservation easements do not. Undeveloped lands were defined as those with housing densities lower than exurban densities (<0.063 units/ha) and not in agricultural or commercial classes as defined for 2001 by Homer et al. (2004). Housing density, land allocation, and proportion developed were all used to classify PACEs into categories of land use change using statistical clustering analysis (Davis and Hansen 2011). From least developed to most developed, the classes were wildland protected, wildland developable, agriculture, exurban, and urban.

The percentage of exotic species in the flora was represented by the proportion of vascular plant species that were nonnative. These data were derived from the NPS Inventory and Monitoring Program's NPSpecies database. NPSpecies is a compilation of species lists and evidence records of species occurrence for vertebrates and vascular plants within national parks. The data are quality checked and certified by subject-matter experts. In addition to reporting numbers of native and nonnative vascular plant species, we used analysis of variance and the Tukey HSD multiple range test to determine if the proportion of vascular plants that were nonnative differed among land use typology classes. Because NPSpecies data are only collected within park boundaries, our analyses deal with the portion of the PACE that is within a national park.

For change in climate over the period 1895–2009, we drew on Haas (2010), who analyzed the PRISM climate data set (Daly et al. 2002). PRISM produces a 4-km resolution surface of monthly climate values annually across the United States, spatially interpolated with weather data from meteorological stations. Haas (2010) used this data to estimate 100-year trends in mean annual temperature, mean annual precipitation, and a moisture index derived from precipitation and potential evapotranspiration. The annual averages across each PACE for the period 1895–2009 were calculated and used to derive rates of change on a per 100-year basis. These trends were estimated with the MM-estimate regression method, which calculates robust standard errors using a bootstrapping method. A separate linear regression was fit for each PACE with $P < 0.05$ the cutoff for statistical significance.

Land use, climate, and biome shifts 2010–2100

Future land use was represented as housing density. We used data from a study (Bierwagen et al. 2010) that projected housing density to 2100 under five scenarios consistent with the main storylines of the IPCC SRES (IPCC 2007). The SRES describe population, socioeconomic, and technological trajectories for broad regions of the world. The scenarios modeled by Bierwagen et al. (2010) varied in assumptions about fertility, domestic migration, international migration, household size, and travel time from urban areas. A county-level spatial interaction model was used to represent domestic migration within the context of a cohort-component population-growth model. The forecasted populations in turn drove the number of housing units required in a county. The Spatially Explicit Regional Growth Model (SERGoM; Theobald 2005) then distributed the housing units to 1-ha areas based on past land-use patterns and travel time along roads from urban areas. We calculated the average housing density within PACEs among these scenarios and report the data for 2030, 2060, and 2090.

Projections of climate and potential shifts in vegetation for the coming century were derived from Rehfeldt et al. (2012). That study used 1.75 million data points to relate the geographic distribution of 46 biome types across North America to current climate variables. They then projected potential biome locations based on climate into the future according to the SRES scenarios, using three general circulation models for the decades surrounding 2030, 2060, and 2090. We used the downscaled future climate projections and potential biome maps from this study because of the coverage of the United States, the rigorous methods, the classification accuracy, comparability of results with other studies, and the relevance of the biome classification to vegetation types in national parks and PACEs. We report averages of projected future average annual temperature and precipitation for the models and scenarios used by that study. These runs differed in projected average temperature increases from 1980–1999 to 2090–2099, with a range of 1.8–4.0°C. Averaging among these future scenarios is simply one way of representing the possible future condition that avoids extreme low or high projections. Moreover, the projected vegetation response to climate change was represented by Rehfeldt et al. (2012) as the consensus among these climate models and scenarios. We calculated the proportion of PACE expected to change in biome type (based on climate suitability) between 2010 and 2100.

Cumulative exposure

The relative magnitude of past change in climate, land use, and invasive species for each PACE was calculated as the percentage of the highest value among the PACEs for each variable. These percentages were summed to represent the relative magnitude of the combined exposure to these three components of global change. We illustrated combined exposure and potential impact into the future by depicting the position of each PACE in the space defined by projected change in housing density 2010 to 2030 and by the percentage of the PACE projected to undergo a biome shift by 2030.

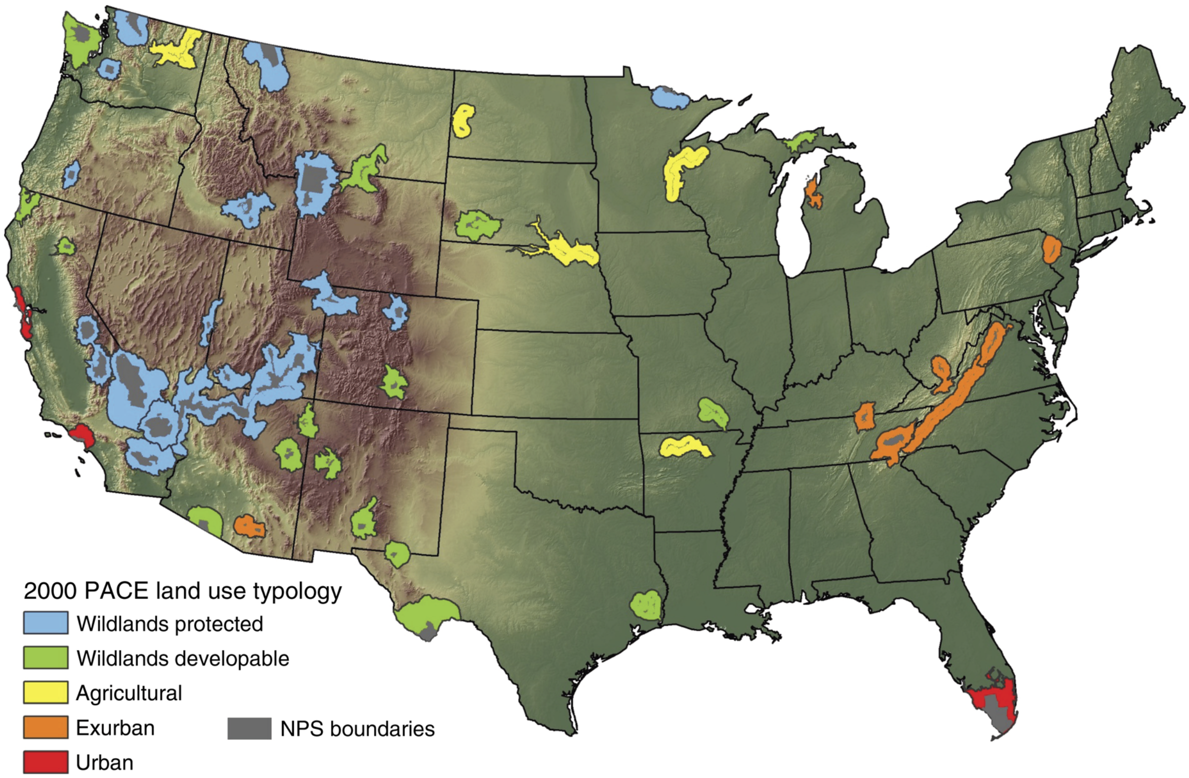

FIG. 2. U.S. National Parks units included in this study. Protected-area centered ecosystems (PACEs) surrounding the park units are color-coded by land use typological membership. Classification criteria were wildland protected, >65% public; wildland developable, <65% public, >60% undeveloped private, <16% agriculture private; agricultural, <65% public, >60% undeveloped private, >16% agriculture private; exurban, <65% public, <60% undeveloped private, <15% private dominated by exurban or urban; urban, <65% public, <60% undeveloped private, >15% private dominated by urban. Park units are listed in Table 1. From Davis and Hansen (2011).

We compared rates of change in housing density for the period 1940–2000 with that projected for 2000–2090 by subtracting the density at the start of each period from the density at the end of the period and dividing by the number of years of the period (e.g., [units/km^2 in 2000 – units/km^2 in 1940]/60 yr = units·km^{-2}·yr^{-1}). The projected future rates were regressed on past rates to determine the slope of the relationship between past and future rates. The relationship was also summarized with correlation analysis. Similar analyses were done for average annual temperature and average annual precipitation. Units for rates of temperature change were °C/100 yr and for precipitation were mm/100 yr.

RESULTS

Exposure 1900–2010

Housing density within PACEs increased on average by 741% from 1940 to 2000 to a mean of 19 units/km^2 (Table 2). By 2000, an average of 27% of the private lands that covered 51% of PACEs had been developed (converted to agriculture, suburban, or urban; Table 1). Individual PACEs differed substantially from these averages (Fig. 3). The median increase in housing density was 224%, which indicates that some PACEs increased well above the average and most PACEs changed less than the average. The North Cascades PACE, for example, decreased in home density since 1940. This PACE is primarily public lands (91%) and 74% of private lands remained undeveloped in 2000. Similarly, the Great Basin PACE in Nevada is only 7% private land, 96% of which remained undeveloped in 2000, and housing density remained low in 2000 (0.03 units/km^2). The Shenandoah PACE near Washington, D.C., in contrast, was heavily subjected to human land use. This PACE is 84% private lands and 80% of these private lands were developed in 2000. Housing density increased by 405% during 1940–2000 to a density of 22 units/km^2. Santa Monica Mountains near Los Angeles, California, had 421 housing units/km^2 in 2000 and an increase of 199% since 1940. The percentage of PACEs in each land use change typology category in 2000 were wildland protected (35%), wildland developable (33%), agriculture (10%), exurban (16%), and urban (6%; Table 1).

Nonnative species represented 13.6% of the vascular plant flora on average within the parks included in the

TABLE 2. Change in housing density and climate during the past century within PACEs and current presence of nonnative vascular plants in parks.

PACE code	Housing units, 2000 (no./km^2)	Change in housing units, 1940–2000 (%)	Nonnative species (no.)	Native species (no.)	Nonnative proportion of total species (%)	Temperature change, 1895–2007 (°C/100 yr)	Precipitation change 1895–2007 (mm/100 yr)	Moisture index change 1895–2007 (mm/100 yr)
ARCH	0.38	1628	58	402	12.6	1.3	0	−1.84
BADL	0.59	27	68	345	16.5	1.1	67.7	0.23
BIBE	0.31	81	92	1270	6.8	1.0	0	0
BICA	0.78	351	111	626	15.1	0.9	0	−1.48
BISO	6.96	169	100	981	9.3	0	0	0.05
BITH	13.33	423	131	1186	9.9	0	250.2	0.90
BLRI	25.19	281	275	1328	17.2	0	0	0
BUFF	5.66	224	221	1132	16.3	0	158.7	0
CACH	1.60	566	107	709	13.1	1.1	0	0
CORI†	3.50	9806	312	2490	11.1	1.4	0	−1.43
CRLA	0.33	−47	43	528	7.5	1.0	0	0
CRMO	0.36	55	88	581	13.2	1.1	0	0
DEVA	0.90	125	53	793	6.3	1.4	44.5	0
DEWA	38.60	250	375	1021	26.9	1.1	0	0
DINO	0.52	803	74	626	10.6	1.1	0	0
ELMA	0.86	3265	32	488	6.2	1.0	0	0
EVER	56.56	2853	382	974	28.2	0.9	0	−1.48
GLAC	0.89	247	129	1039	11.0	1.4	0	0
GRBA	0.03	−33	60	673	8.2	0.7	0	0
GRSA	0.78	35	24	563	4.1	0	0	0
GRSM	17.73	501	339	1280	20.9	0	0	0
GUMO	0.13	55	11	969	1.1	0.7	0	−2.14
JOTR	8.68	1836	24	484	4.7	0.6	0	0
LARO	4.08	273	119	500	19.2	0.9	0	0
LAVO	2.41	223	41	720	5.4	0.9	0	0
MNRR	4.21	157	84	411	17.0	0.7	84.2	0
MOJA	0.41	424	84	820	9.3	0.8	0	0
MORA	3.78	−21	145	775	15.8	0.8	0	0
NERI	13.78	81	206	978	17.4	0	0	0
NOCA	0.60	−70	227	1173	16.2	1.2	0	0
OLYM	11.16	325	226	967	18.9	0.5	0	0
ORPI	0.22	816	61	623	8.9	1.3	0	0
OZAR	4.66	111	75	803	8.5	0	132.2	0
PEFO	0.49	231	54	428	11.2	1.4	0	0
PIRO	1.19	78	119	773	13.3	1.0	67.2	0
POGO	179.14	143	379	808	31.9	1.4	0	0
REDW	3.90	296	219	599	26.8	0.7	0	0
ROMO	2.41	194	83	918	8.3	1.4	0	0
SACN	17.14	181	169	1289	11.6	1.2	0	−1.69
SAGU	35.15	8018	99	1138	8.0	0.8	105.0	0.07
SAMO	421.20	199	306	836	26.8	1.4	0	0
SHEN	21.67	405	348	1040	25.1	0.4	82.0	0
SLBE	8.99	236	221	875	20.2	0	163.6	0.14
THRO	0.63	124	62	455	12.0	1.5	0	0
VOYA	1.25	−45	141	870	13.9	0.9	49.3	0
WHSA	1.86	1948	15	287	5.0	0.8	53.4	0
YELL	0.45	213	219	1651	11.7	1.1	0	0
YOSE	1.36	33	253	1734	12.7	1.0	0	−0.31
ZION	1.70	1172	97	645	13.1	1.8	0	−0.48
Mean	18.95	741	146	869	13.6	0.9	25.7	−0.2

Notes: For climate, data are statistically significant trends during 1895 to 2007; nonsignificant trends are denoted with 0. Climate data are from Haas (2010).

† Colorado River parks are: Canyonlands, Capitol Reef, Glen Canyon, Grand Canyon, and Lake Mead.

study (Table 2). Some parks had less than 5% nonnative plant species (e.g., Guadalupe Mountains, Great Sand Dunes, and Joshua Tree; all in the southwest deserts of the United States; Fig. 3). In contrast, nonnative plant species make up 28% of the vascular flora in the Everglades and Big Cypress parks near Miami, Florida, and 32% in the Point Reyes and Golden Gate parks near or in San Francisco, California. Presence of nonnatives was related to the land-use groupings above. Wildland PACEs had the lowest percentage of nonnatives within parks (11%) and urban PACEs the highest (29%; Fig. 4). The percentage of nonnative plants was significantly higher in parks in urban PACEs than all other groups, and significantly higher in parks in exurban PACEs than parks in the wildland groups.

PACEs warmed by an average of 1.0°C/100 yr since 1895 (Table 2). The highest rates of warming (1.4° to

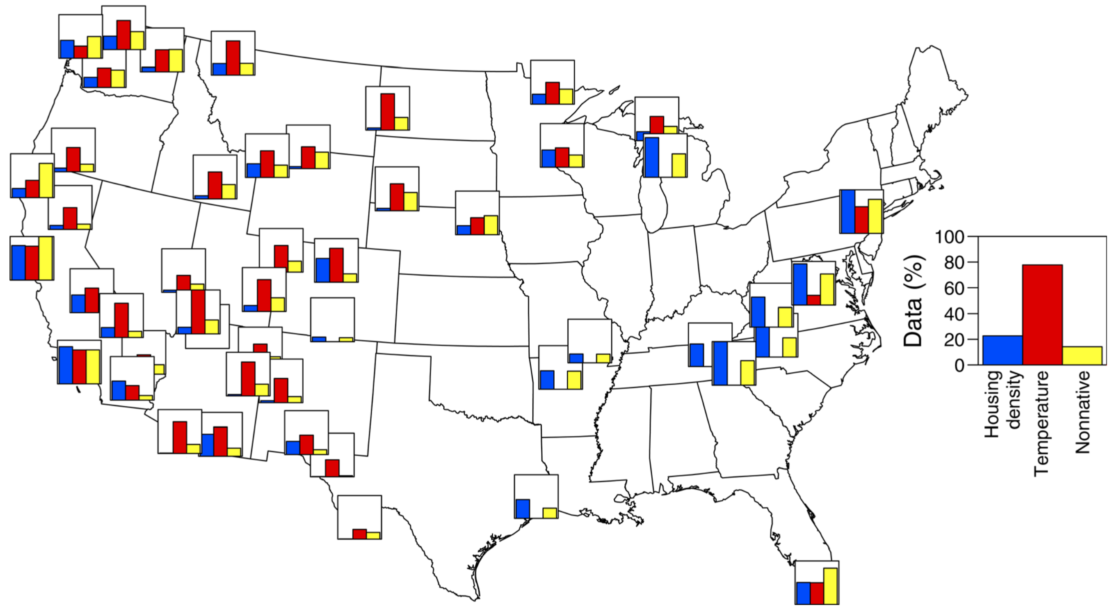

Fig. 3. Relative change in housing density 1940–2000, change in temperature 1900–2000, and percentage of total vascular plant species that was nonnative based on data from 2010. Values are normalized to the value of the PACE with the highest level of change. For housing density, negative values were changed to 0 and percent change is expressed as $\log(x+1)$, where x is the percent change in housing density, to reduce skew due to a few PACEs with very high levels of increase in housing density. The inset is a legend showing the x- and y-axes on the bar graphs that are depicted in the map.

1.8°C/100 yr) occurred in the Southern Rockies and Colorado Plateau region in PACEs including Zion, Rocky Mountain, Petrified Forest, and Lake Mead (Fig. 3). No significant warming occurred in several PACEs, mostly in the eastern United States such as Great Smoky Mountains, Blue Ridge Parkway, and Ozark. Precipitation increased significantly in 22% of the PACEs, largely in the midwest United States. The moisture index revealed that 17% of the PACEs increased in water balance while 3% decreased, and 80% were unchanged.

Combining the relative magnitude of changes in climate, land use, and invasive species revealed the wide variation in exposure to these elements among the PACEs (Fig. 3). Individual PACEs had relatively high rates for one, two, or all three elements of exposure. Three PACEs, Santa Monica Mountains, Point Reyes-Golden Gate, and Delaware Water Gap had normalized rates of change more than five times those in Great Sand Dunes and other western interior PACEs.

Exposure and potential impact 2000–2100

Projected increases in housing density from 2000 averaged across PACEs and four IPCC SRES scenarios was 42% by 2030, 125% by 2060, and 255% by 2090 (Table 3). Several PACEs, largely in the midwest United States, had projected housing density increases of less than 10% in 2090 (Fig. 5). In contrast, PACEs largely in the southwestern United States were projected to increase in housing density by 90–600% by 2030 and 450–4300% by 2090.

Projected future temperature trends averaged among the climate models and IPCC scenarios indicated that PACEs may warm by 0.9 to 2.4°C (mean = 1.76°C) by 2030 (Table 3). By 2090, mean annual temperatures are projected to be 2.5°–4.5°C (mean = 3.7°C) warmer than present. PACEs with the highest projected warming rates are in the southwest deserts and western mountains. Projected temperature increases by 2030 in these locations are up to twice as great as in PACEs in the eastern and midwestern United States (Fig. 5). Mean annual precipitation is projected to increase in all but

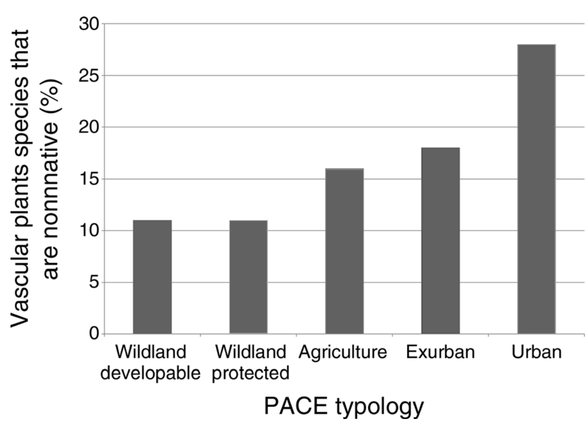

Fig. 4. Percentage of vascular plant species that are nonnative within each type of protected-area centered ecosystem (PACE). The two wildland classes differ significantly from the exurban and urban classes (ANOVA, $N = 48$ sites, $F_{4,44} = 10.68$, $P < 0.001$).

TABLE 3. Projected future change in housing density, climate, and biome suitability.

PACE Code	Change in housing units (%)			Change in temperature (°C)			Change in precipitation (mm)		
	2000–2030	2000–2060	2000–2090	2000–2030	2000–2060	2000–2090	2000–2030	2000–2060	2000–2090
ARCH	6.39	12.78	19.61	1.88	2.72	3.68	11.34	49.79	42.34
BADL	5.39	11.79	18.22	1.72	2.91	4.36	5.47	8.63	−5.18
BIBE	3.18	8.36	14.29	1.66	2.31	3.11	−1.55	64.69	82.95
BICA	0.99	1.98	2.97	1.77	2.81	3.73	14.05	39.72	32.90
BISO	1.41	3.03	4.65	1.60	2.49	3.69	21.95	131.67	125.81
BITH	12.4	33.5	63.99	1.73	2.56	3.44	13.91	−110.96	5.85
BLRI	5.95	13.13	22.12	1.46	2.26	3.54	35.87	141.89	138.00
BUFF	1.88	3.78	5.68	1.96	2.94	4.04	1.96	−48.73	23.72
CACH	22.27	52.71	84.34	2.06	2.72	3.70	18.55	55.08	63.04
CORI†	120.47	340.45	666.82	2.42	3.36	4.21	11.38	74.04	92.93
CRLA	2.13	4.26	6.39	2.07	3.27	4.28	−6.28	−83.21	−200.65
CRMO	14.6	32.82	52.34	1.94	3.23	4.05	3.69	18.34	6.14
DEVA	132.17	400.35	840.04	2.22	3.42	4.16	6.54	56.42	69.71
DEWA	44.24	134.08	303.91	1.26	2.44	3.76	51.66	81.40	151.82
DINO	6.6	13.34	20.49	1.93	2.79	3.82	4.42	48.11	22.89
ELMA	9.81	20.88	32	2.13	2.59	3.60	19.37	49.39	59.92
EVER	59.1	179.12	374.84	0.91	1.63	2.50	−16.23	11.72	−21.23
GLAC	19.06	38.12	57.18	1.45	2.95	3.72	22.96	19.49	31.60
GRBA	7.67	24.28	46.26	2.08	3.16	4.09	4.44	94.19	87.38
GRSA	21	44.26	68.15	2.24	2.55	3.82	4.62	37.83	21.29
GRSM	12.53	26.88	43.07	1.85	2.59	3.79	20.98	143.63	157.65
GUMO	10.48	30.93	51.48	1.87	2.23	2.98	−5.85	68.97	82.31
JOTR	89.99	243.06	452.08	1.78	2.74	3.36	13.26	73.75	134.26
LARO	11.18	28.94	55.21	1.73	3.32	4.24	8.74	−29.74	−47.76
LAVO	10.13	23.91	40.75	2.13	3.46	4.39	1.93	−65.50	−166.89
MNRR	2.13	5.41	9.68	1.75	3.02	4.65	18.87	−29.87	−24.58
MOJA	608.78	1993.59	4227.69	1.36	2.41	3.09	12.10	105.89	173.37
MORA	44.91	114.41	215.43	1.79	3.09	4.05	27.30	−192.47	−312.18
NERI	0.12	0.35	0.62	1.49	2.38	3.68	37.60	134.16	112.45
NOCA	24.13	49.71	76.25	1.71	3.09	4.07	42.20	−145.70	−215.11
OLYM	16.36	38.14	67.7	1.64	2.48	3.50	28.05	−341.14	−441.58
ORPI	189.04	579.81	1190.01	1.67	2.45	3.04	13.17	9.20	143.58
OZAR	0.47	0.98	1.49	1.82	2.90	4.07	15.04	−17.99	43.07
PEFO	18.83	57.37	108.5	2.05	2.64	3.60	20.52	41.40	59.13
PIRO	0	0	0	1.24	2.33	3.52	28.14	16.84	70.56
POGO	14.44	38.97	77.31	1.52	2.53	3.25	22.67	−58.39	−183.89
REDW	5.41	10.82	16.23	1.93	2.98	4.00	2.22	−139.54	−316.09
ROMO	121.88	333.05	667.46	2.03	2.64	3.92	10.64	48.97	20.76
SACN	28.96	78.58	153.78	1.52	2.61	3.97	23.31	−4.06	37.59
SAGU	37.54	105.08	208.01	1.77	2.39	3.12	30.96	35.00	131.71
SAMO	27.78	76.12	144.86	1.63	2.69	3.34	6.69	9.06	−3.86
SHEN	26.5	68.63	127.23	1.37	2.34	3.67	38.22	107.20	106.63
SLBE	9.39	20.2	33.34	1.32	2.38	3.45	34.79	−4.05	54.66
THRO	0.78	1.66	2.54	1.46	2.74	3.85	15.88	9.47	22.88
VOYA	0	0	0	1.35	2.43	3.58	21.25	18.91	62.76
WHSA	21.4	54.86	97.28	1.98	2.28	3.13	14.32	47.45	60.38
YELL	30.03	60.87	91.71	1.85	3.01	3.89	18.23	64.13	36.87
YOSE	107.12	349.02	752.08	2.08	3.68	4.46	13.51	−26.11	−87.60
ZION	110.93	389.84	882.57	2.04	3.01	3.91	15.67	130.91	134.72
Mean	42.41	125.6	255.08	1.76	2.73	3.73	15.89	15.30	13.82

Notes: Housing density projections were based on the average of four IPCC future growth scenarios (A1, A2, B1, and B2). Climate change projections were based on the average of seven climate change model scenarios as downscaled by Rehfeldt et al. (2012). Biome suitability is the proportion of each PACE that is projected to undergo a shift in biome suitability based on consensus among six climate models scenarios by Rehfeldt et al. (2012).

† Colorado River parks are: Canyonlands, Capitol Reef, Glen Canyon, Grand Canyon, and Lake Mead.

four PACEs by 2030, most substantially in the eastern and upper midwestern United States. Projections for 2090 indicate a reduction in precipitation in 13 PACEs, with the greatest reductions in the southwest deserts and the greatest increases in other parts of the United States.

On average, 30% of the area within PACEs are projected to experience climates unsuitable for current biomes by 2030 and 40% by 2090 (Table 3). Some 15 PACEs, mostly in the upper midwestern and eastern United States, are projected to experience climate-related biome shifts in less than 5% of their areas by 2090 (Fig. 5). In contrast, 14 PACEs in the mountain and southwestern United States are projected to experience unsuitable climates for their present biome types across 50–86% of their areas by 2030 and up to 96% by 2090. It is places with high projected climate change and places with topographic complexity where climate-driven biome shifts are projected to be most

TABLE 3. Extended.

Biome suitability change (proportion)		
2000–2030	2000–2060	2000–2090
0.54	0.56	0.56
0.00	0.00	0.00
0.40	0.43	0.50
0.45	0.60	0.66
0.00	0.00	0.00
0.07	0.10	0.20
0.00	0.01	0.01
0.03	0.03	0.03
0.50	0.53	0.69
0.39	0.51	0.56
0.26	0.62	0.64
0.56	0.71	0.64
0.15	0.20	0.20
0.00	0.00	0.00
0.61	0.65	0.65
0.27	0.31	0.33
0.57	0.56	0.61
0.57	0.81	0.88
0.56	0.64	0.70
0.70	0.79	0.73
0.01	0.02	0.01
0.31	0.31	0.33
0.39	0.56	0.56
0.17	0.24	0.26
0.26	0.29	0.36
0.09	0.09	0.09
0.30	0.54	0.46
0.29	0.51	0.46
0.01	0.01	0.01
0.57	0.60	0.63
0.22	0.22	0.40
0.00	0.00	0.00
0.05	0.05	0.05
0.88	0.88	0.88
0.00	0.00	0.00
0.69	0.87	0.96
0.39	0.42	0.55
0.65	0.77	0.86
0.12	0.17	0.12
0.23	0.29	0.33
0.52	0.55	0.63
0.01	0.01	0.01
0.00	0.00	0.00
0.00	0.00	0.00
0.00	0.34	0.00
0.40	0.49	0.49
0.43	0.60	0.89
0.34	0.46	0.49
0.63	0.74	0.83
0.30	0.37	0.39

prevalent (e.g., Glacier, Greater Yellowstone, and Rocky Mountain in the Rocky Mountain region and Petrified Forest in the southwestern deserts).

Combined potential exposure to land use change and potential impact of climate change can be represented as the magnitude of projected climate-driven biome shift and projected changes in housing density by 2030 (Fig. 6). Some PACEs are expected to experience little change in either metric (e.g., Painted Rocks, Voyagers, Theodore Roosevelt, Organ Pipe Cactus). Others are high in either housing density (e.g., Delaware Water Gap, Great Smoky Mountains, Saguaro) or potential climate impact (e.g., Petrified Forest, Great Sand Dunes), and a few PACEs are high in both land use intensity and potential climate impact (e.g., Point Reyes/Golden Gate, Santa Monica Mountains, Rocky Mountain).

Past and potential future exposure

The average rate of change in housing density (units$\cdot$km$^{-2}\cdot$yr^{-1}) among the four IPCC scenarios in the coming century was highly correlated with the actual rate of change during 1940–2000 (correlation coefficient = 0.97). The slope of the relationship between change in units/km^2/year during 1940–2000 and 2000–2090 was 0.82, indicating that on average projected future change in home density is 82% of the past rate. PACEs with projected future rates well below the past rate include those with relatively fast rates of increase in the past (Santa Monica Mountains, Point Reyes/Golden Gate, Shenandoah, Blue Ridge, Great Smoky Mountains; Fig. 7). PACEs projected to increase more rapidly in housing density in the future than in the past include Everglades and Delaware Water Gap, both near major cities, and several PACEs in the south and west United States, such as Joshua Tree, Colorado River, Zion, Death Valley, Mojave Desert, Rocky Mountain, Yosemite, and Mount Rainier. The magnitude of change in housing density during the 1940–2090 reference period can be represented as number of PACEs in recognized land use classes at the beginning of the time period (Table 4). The number of PACEs with undeveloped and very low housing density levels dropped from 38 to 22 during this period, the number in the exurban class increased from 4 to 18, and the number of PACEs in the urban/suburban class increased from 1 to 3.

Unlike housing density, there was virtually no relationship between change in temperature in the last century and projected change in the coming century (correlation coefficient = 0.07). All PACEs were projected to increase in temperature during 2000–2100 (by 2.5°–5.8°C), regardless of change during 1895–2007 (Fig. 8). Total past and projected change in temperature for 1900–2100 ranged from +3.7°C in the Everglades and Big Cypress PACE to 6.1 in the Zion PACE. A total of 27 PACEs had total projected changes of >5°C for this period.

Given that relatively few PACEs experienced significant changes in annual precipitation in the past century, total past and projected future change 1900–2100 was mostly a result of projected change in the coming century (Fig. 9). The total change ranged from –490 to 0 mm per 100 yr for 11 PACEs largely on the West Coast to 0 to +257 mm per 100 yr for 38 PACEs scattered widely across the United States.

DISCUSSION

The results revealed that these U.S. parks and surrounding PACEs have, on average, undergone relatively high rates of exposure to regional and global change during the last century. Housing density, one measure of land use intensity, increased by nearly 750%

during 1940–2000. Of the 51% of PACE area that is private land, nearly 28% has been converted to agriculture, exurban, suburban, or urban land uses. Nearly 14% of the vascular plant flora within the parks is comprised of nonnative species. Average annual temperature has warmed 1.0°C/100 yr since 1895. This level of exposure to human-induced change raises important and still outstanding questions as to whether these parks are functioning as "natural" systems.

The central assumption of the PA approach to conservation is that restricting human activities within protected areas will allow natural processes and native species to persist (Gaston et al. 2008). We know relatively little, however, about the magnitude of human influence that results in loss of ecological function and native species. Increasing human densities and land development can influence PAs through reducing ecosystem size, changing flows of materials and disturbances, reducing crucial habitats, and increasing negative human edge effects (Hansen and DeFries 2007). The net results of such changes include fragmentation of natural habitats (Piekielek and Hansen 2012), reduction in connectivity with other PAs (Berger 2004), loss of essential natural disturbance regimes (Baker 1992), destruction of population source habitats (Hansen 2011), and extinction of native species. For example, prior study of western parks found that a substantial amount of the variation in native species extinction rate was explained by human density in the surrounding area (Parks and Harcourt 2002). These studies indicate that some U.S. parks have undergone ecological degradation since establishment, partially due to exposure to the types of human influences documented here.

U.S. National Parks and PACEs will likely be further challenged by human-induced change in the coming century. The projections summarized in this study suggest that housing densities will continue to increase in PACEs, at about 82% of the rate of past decades. The rate of climate warming in the coming century is projected to be 2.5–5.8 times higher than that measured in the past century. A potential impact of this change in climate is that some 40% of the area within PACEs will experience climates unsuitable for current biomes by 2090.

The ecological responses in the parks in the coming century will reflect the combined exposure of the past decades and future decades. The magnitude of change in measured or projected PA exposure for the 200 years from 1900 to 2100 is dramatic. Home density across PACEs increases from 7.4 units/km^2 in 1940 to 48.3 units/km^2 in 2090. Whereas 5 PACEs had home densities in the exurban or urban/suburban classes in 1940, 21 are projected to be in these classes by 2090. Mean annual temperature increases on average 5°C. To put this into context, the change in mean July temperature across North America since the end of the last ice age 14 000 years ago varied ~5°C (Viau et al. 2006). Globally, projected temperature increases by 2100 exceed those observed during the past 11 300 years (Marcott et al. 2013).

While these average rates of exposure to human-induced change and potential impact are sobering, they mask important variability among PACEs. Some PACEs have decreased or changed little in housing density since 1940, have relatively few nonnative vascular plant species, and/or have changed little in climate since 1900. While all PACEs are projected to warm in the coming century, increased moisture and other factors result in future climate conditions that continue to be suitable for current biomes in most PACEs east of the Rocky Mountains. Other PACEs have changed and are projected to change substantially in one or all of these factors. For example, Point Reyes-Golden Gate near San Francisco, California, has a projected increase in housing density of 247% from 1940 to 2100, an increase in mean annual temperature of 4.5°C for 1900–2100, a decrease in mean annual precipitation of 204 mm for 1900–2100, and 96% of the PACE is projected to have climate unsuitable for current biome vegetation by 2090. Such variation among PACEs is a result of geographic variation in land use, climate change, and sensitivity of ecosystems.

Patterns of land use are known to vary with proximity to resources, markets, past development, and natural amenities (Huston 2005). Within the United States, Euro-American settlement was initially associated with areas high in natural resources (e.g., farm lands) and transportation corridors, and thus focused on the seacoasts, the eastern United States, and eventually the Midwest (Huston 2005). Improvements in transportation technology during ca. 1850–1950, fueled expansion of settlement into the interior United States along rail lines and highways and to locations with airports. Changes in job opportunities, improved transportation, and increased wealth lead to rapid growth of cities, expansion of suburbs, and a decline in population density in many rural areas during 1950–1970 (Brown et al. 2005). Since 1970, a "rural rebound" has been in place where locations high in natural amenities have attracted rapid exurban expansion and growth of small cities. Accordingly, PACEs in the eastern United States and West Coast, experienced rapid land use intensification during 1940–1970 while those in the High Plains, Rocky Mountains, and Desert Southwest generally remained low in human development. Since 1970 and especially in the 1990s, rural and wilderness PACEs with high natural amenities have undergone rapid increases in exurban development. The future projections for housing density based on the average of the SRES suggest that the southwest United States will be the region of highest increases.

Rates of climate change vary with latitude, distance from oceans, and topography (IPCC 2007). The ecological impacts of this past climate change are known to vary geographically across the United States. Climate warming and drying has been particularly pronounced

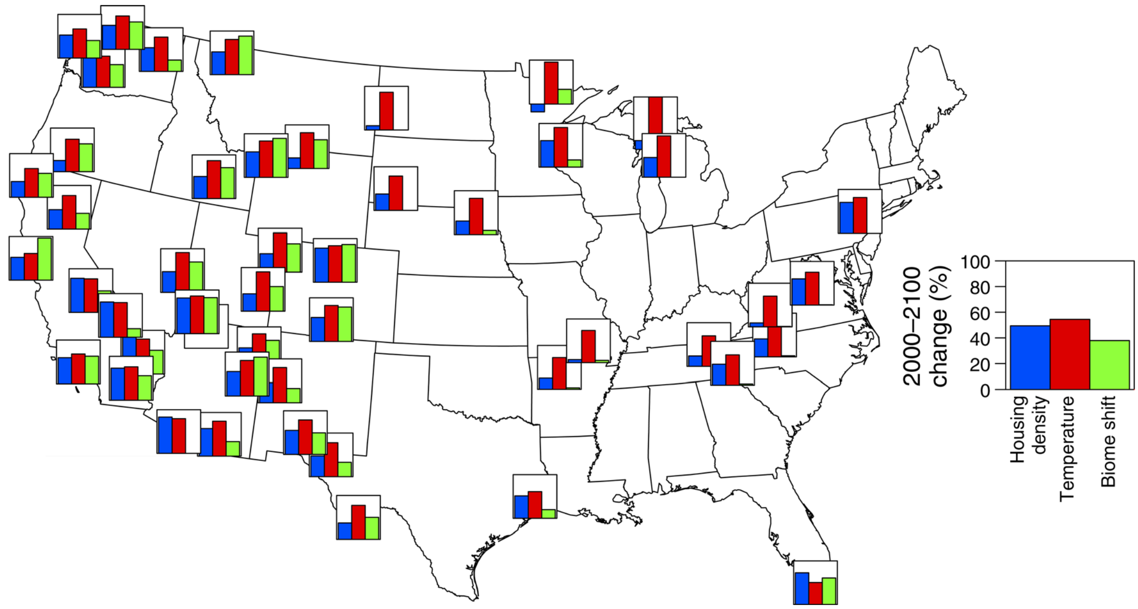

FIG. 5. Relative change in projected housing density, temperature, and percentage of PACE undergoing a shift in biome type suitability for 2000–2100. Housing density and temperature are normalized as in Fig. 3. Housing density is the average of the predictions of four scenarios (Bierwagen et al. 2010). Temperature is based on the average of six climate models and scenarios (Rehfeldt et al. 2012) and biome suitability is under the consensus of six climate models and scenarios (Rehfeldt et al. 2012). The inset is a legend showing the *x*- and *y*-axes on the bar graphs that are depicted in the map.

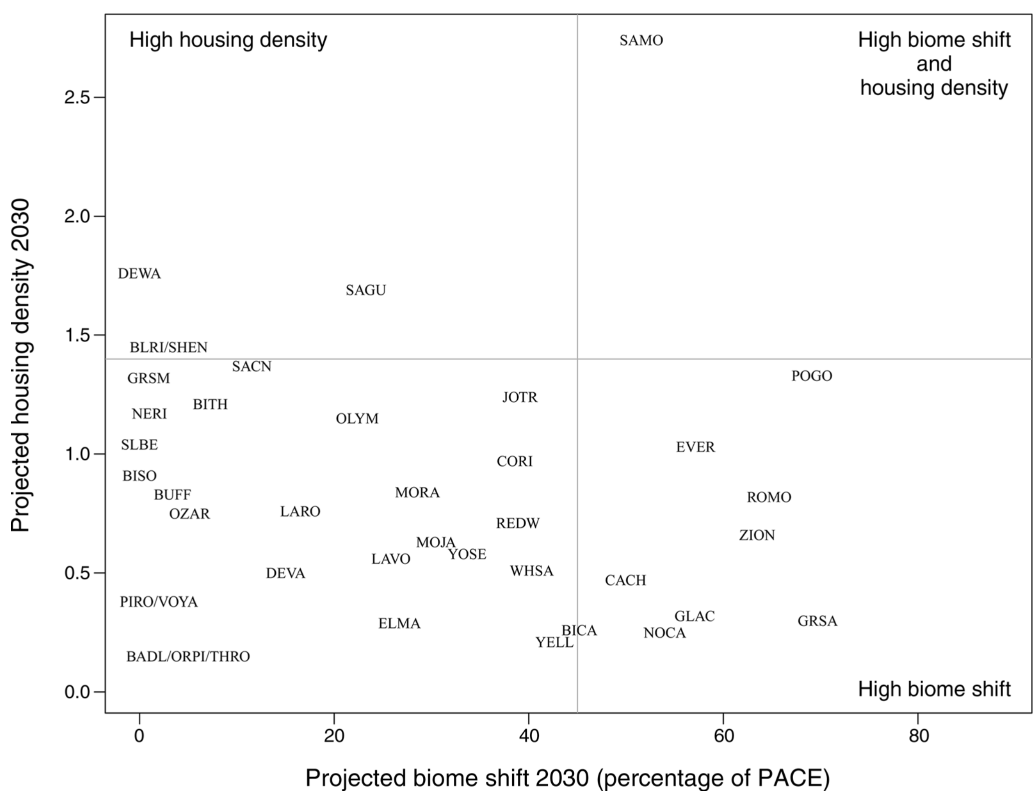

FIG. 6. Distribution of PACEs in bivariate space of housing density vs. biome shift projected for 2030. Scenarios are described in the Fig. 5 legend. Housing density has been transformed as $\log([\text{no. houses/km}^2] + 1)$. PACE codes appear in Table 1.

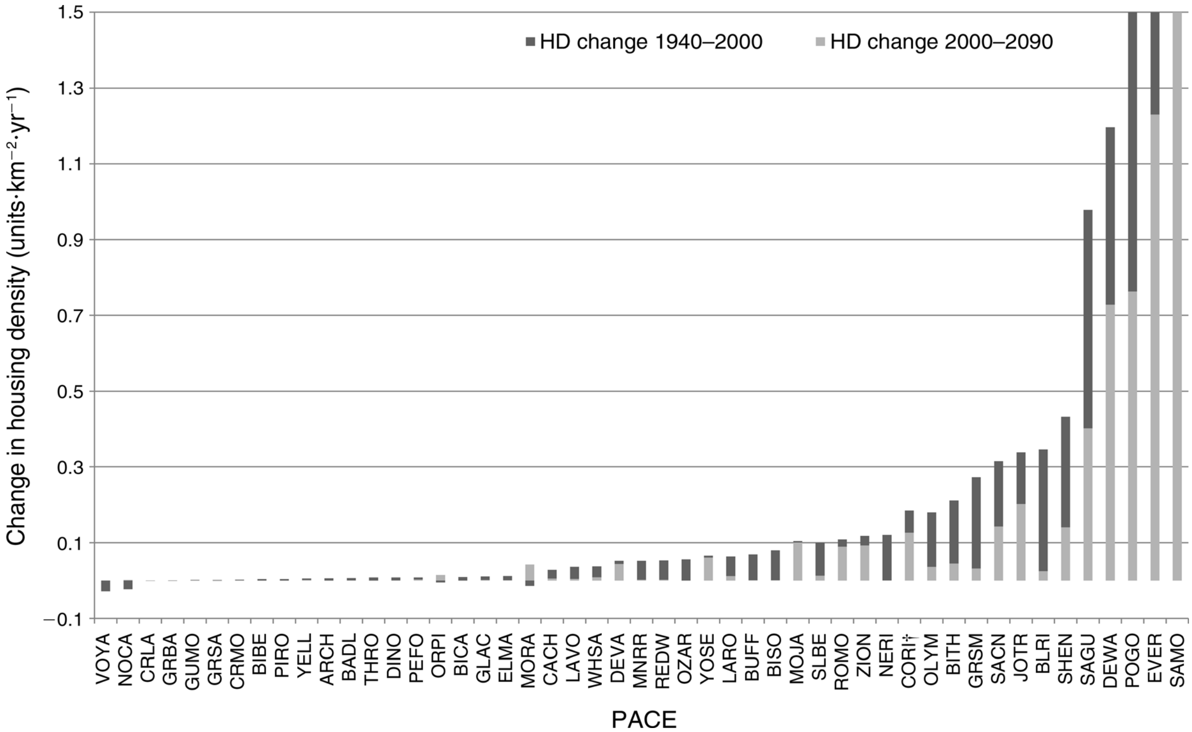

FIG. 7. Rates of change in housing density (HD, units·km^{-2}·yr^{-1}) projected for 2000–2090 relative to observed rates during 1940–2000 for PACEs included in the study. The y-axis was constrained to a maximum value of 1.5 units·km^{-2}·yr^{-1} to better display differences among most PACEs. The actual values for the SAMO PACE were 3.9 units·km^{-2}·yr^{-1} for 1940–2000 and 3.2 units·km^{-2}·yr^{-1} for 2000–2090.

within western states, resulting in increased frequency of severe fires, widespread forest pest outbreaks, and drought-induced forest mortality (Westerling et al. 2006, Allen et al. 2010). These factors in combination have led to large scale forest die-off especially in the southwestern deserts, the Rocky Mountains, and the Sierra Nevada (Breshears et al. 2005). Some keystone tree species are at risk of extinction in the United States due to these changes. Mature whitebark pine (*Pinus albicaulis*) has undergone very high levels of mortality in the past decade in the Greater Yellowstone Ecosystem and was recently designated as a candidate listing as a threatened or endangered species (U.S. Fish and Wildlife Service 2011). This species provides a critical food source for the grizzly bear (*Ursus arctos*), which was recently relisted as endangered, in part because of concern of the loss of whitebark (U.S. Fish and Wildlife Service 2010). Piñon pine (subgenus *Ducampopinus*), saguaro cactus (*Carnegiea gigantea*), and Joshua tree (*Yucca brevifolia*) have all experienced high mortality in multiple western parks (Saunders et al. 2009). Future projections of climate change relative to the climate associations of biomes suggest that PACEs throughout the western United States will undergo the largest loss in area of climates suitable for current biomes.

Cumulative and synergistic effects

The elements of human-induced change, including land use, exotic species, and climate, often do not influence ecological systems in isolation. Their effects may be cumulative or synergistic. Cumulative effects are changes to the environment that are caused by an action in combination with other past, present and future human actions (e.g., Canadian Environmental Assess-

TABLE 4. The number of PACEs in each of four categories of housing density in 1940 and projected for 2090 under the average of four IPCC scenarios.

Category	Housing density (units/km^2)	Number of PACEs in class in 1940	Number of PACEs projected for class in 2090
Undeveloped/very low density	0–3.1	38	22
Rural	3.1–6.3	6	6
Exurban	6.3–145	4	18
Urban/suburban	>145	1	3

Note: Housing density categories are from Davis and Hansen (2011).

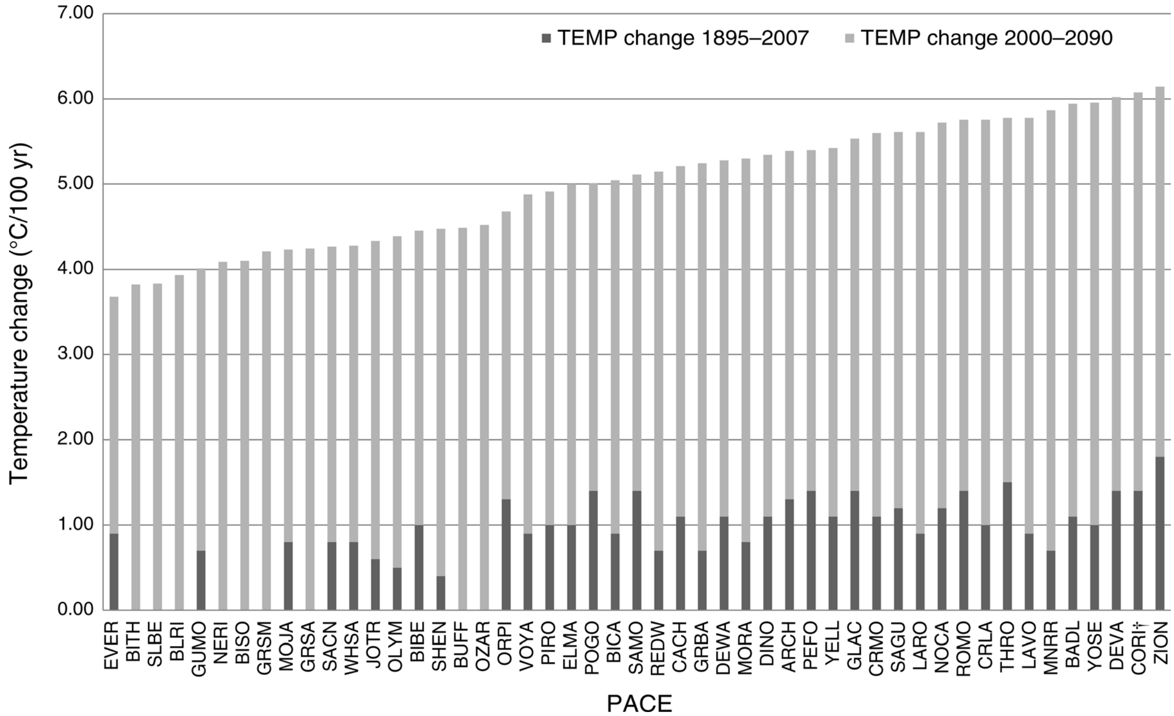

FIG. 8. Rate of change in average annual temperature (TEMP, °C/100 yr) during 1895–2007 and the projected rate for 2000–2090 for PACEs included in the study.

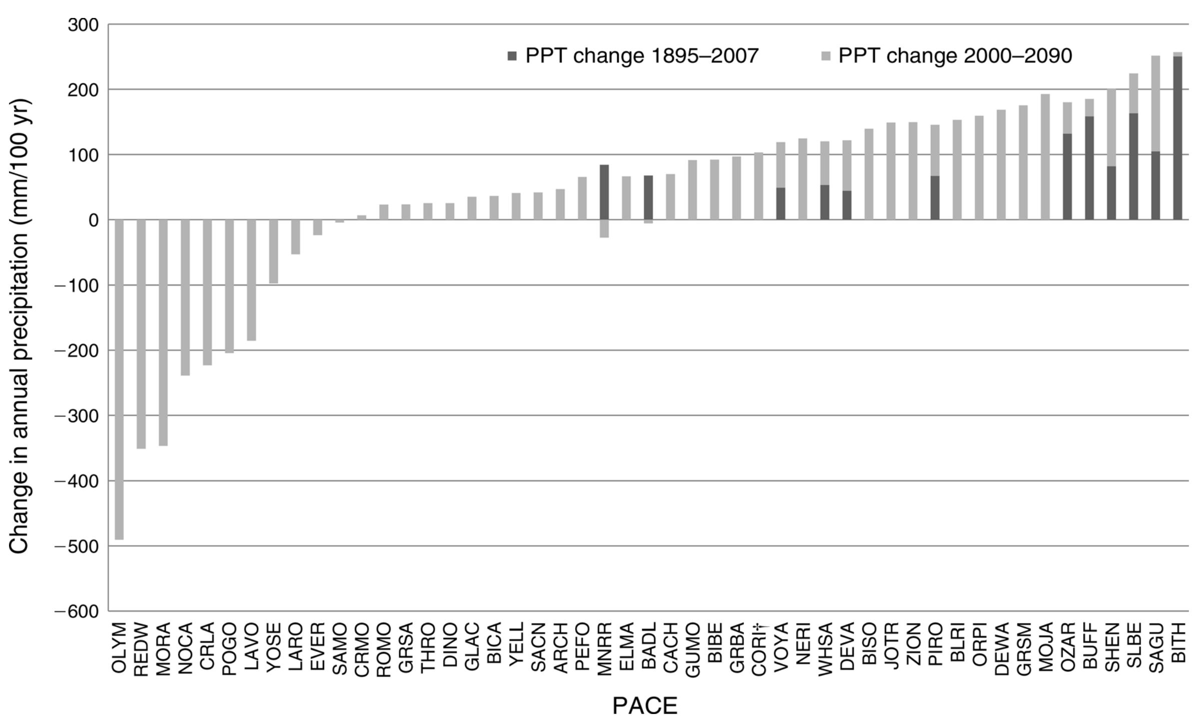

FIG. 9. Rate of change in average annual precipitation (mm/100 yr) during 1895–2007 and projected rate for 2000–2090 for PACEs included in the study.

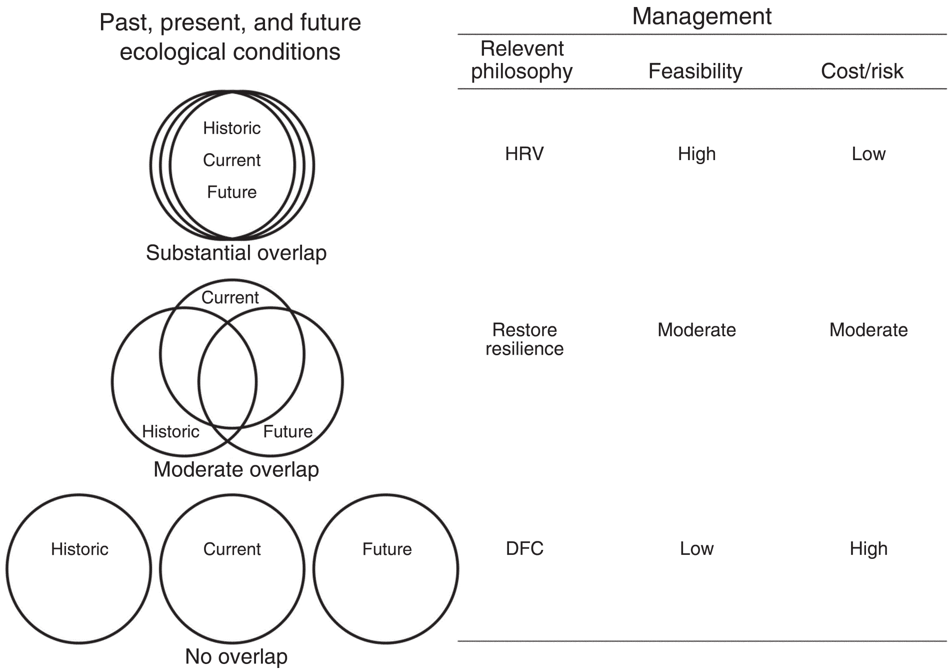

Fig. 10. Illustration of three possible cases of the extent to which current ecosystem conditions in a place differ from historic conditions and from projected future conditions. Circles denote the range of variability for each time period. Also shown is the expected management criteria for each case. Abbreviations are HRV, historic range of variability and DFC, desired future conditions.

ment Agency 1999). Accordingly, the effects of change in land use, exotic species, and climate on PAs may be additive. For example, reductions in habitat area for a species often reflect the additive effects of habitat loss due to climate change and due to land use intensification (Jetz et al. 2007). Accordingly, future analyses of U.S. National Parks and PACEs could best quantify changes in habitat area by overlaying losses due to changes in climate suitability and land use intensification.

Synergistic effects are those where the effects of two elements of exposure are greater than their additive effects due to interactions between them (e.g., Rosa and Seibel 2008). For example, a meta-analysis of habitat fragmentation (Mantyka-Pringle et al. 2012) found that the negative effects of fragmentation were exacerbated in places of climate-change induced drought stress, resulting in elevated loss of biodiversity. Another example of synergy is where the abilities of species to adapt to climate change is reduced by land use impacts. Habitat loss and fragmentation may increase species susceptibility to climate change by limiting their ability to track climate variations across the landscape. Synergies are also expected among climate change, land use, and biotic invasions. Both climate and land use change may favor biological invasions and enhance negative impacts on ecosystem processes and native species as is the case with cheatgrass (*Bromus tectorum*) invasion in the Great Basin (Bradley 2010). Quantifying the combined exposure of PACEs to change in land use, nonnatives, and climate is important to identify which PACEs may be most vulnerable to cumulative and synergistic effects.

Among the PACEs included in this study, Point Reyes-Golden Gate, Santa Monica Mountains, and Delaware Water Gap had the highest combined exposure to land use intensification, nonnative plants, and temperature increase over the past century. The Santa Monica Mountains PACE, for example, is surrounded by the greater Los Angeles, with some 72% of the PACE being developed in 2000. Nearly one-quarter of the vascular plants are not native. Temperature has warmed by 1.45°C in the past century. Housing density within this PACE is projected to rapidly increase and more than 50% of the PACE is projected to experience a climate unsuited to current biomes by 2030. Potential cumulative effects include loss of species' habitats due to conversion to housing, shifts in climate that exceed species tolerances, and competitive effects of invasive species. Potential synergistic effects include reduced adaptive capacity to disperse to newly suitable habitats due to constraints imposed by urban development, and decreased vigor or fitness due to pollution. Such cumulative and synergistic effects are likely to be most prevalent in the PACEs that have the greatest observed and projected exposure to land use and climate change over the past and in the coming century. Among these PACEs with high exposure are several iconic wilderness PACEs in the Rocky Mountains and desert southwest such as the Rocky Mountain PACE and the Colorado River PACE and more urban coastal PACEs including the Everglades and Big Cypress, Santa Monica Mountains, and Point Reyes and Golden Gate PACEs.

Scope and limitations

This study aimed to demonstrate the value of conducting vulnerability assessments of PAs as a guide to management under human-induced change. For logistical reasons, this analysis dealt with only a subset of the vulnerability assessment approach of Glick et al. (2011). We primarily examined *exposure* (to land use, climate, and exotic species). *Potential impact* was only represented as potential biome shifts based on vegetation sensitivity to climate change.

Future analyses should further consider *sensitivity* of native species and ecological processes, *adaptive capacity*, and *vulnerability* to change in land use, climate, and invasives. It is our hope that this study makes an important contribution that will motivate more complete vulnerability assessments for U.S. National Parks, their surrounding PACEs and protected areas globally. This important work will lay the basis for climate adaptation strategies aimed at maintaining ecological condition of parks under global change.

Additional elements of exposure are relevant to particular PAs. These include air and water pollution, infrastructure development in addition to housing (e.g., roads), and direct human impacts such as those relating to poaching, pets, or recreation. Such additional elements of exposure may also appropriately be considered as criteria for delineating PACEs. The airsheds of sources of air pollution, for example, could be mapped as a basis for mapping PACEs (see Hansen et al. 2011).

More generally, careful attention to designation of PACEs will improve vulnerability assessments. The PACE concept is important because it provides a basis for identifying the areas outside PAs that are relevant to ecological integrity within the PAs and thus should be included in vulnerability assessments. Criteria for mapping PACEs should be carefully selected and research directed at testing the assumptions underlying these criteria. Additionally, sensitivity analyses could be done to determine how, for example, exposure varies with assumptions about the criteria and resulting PACE boundaries (Hansen et al. 2011).

A key limitation to assessment of vulnerability to past and future change is the uncertainty associated with estimates of exposure, potential impact, and vulnerability (Glick et al. 2011). Each of the data sets used in vulnerability assessments have levels of error associated with them. Models that link these data sets have outputs with errors that are multiplicative of those in the individual data sets (see Huntley et al. 2010). Thus, uncertainty in vulnerability assessment may be high. Resource managers are increasingly cautious about uncertainty and methods to include uncertainty in decision making have been developed (Peterson et al. 2003). In the case of this study, few of the data sets used had been validated nor error level quantified. The data sets used for exposure over the past century are all based on empirical observations (e.g., census surveys of homeowners, climate data collected at meteorological stations) with stringent quality control procedures. However, some level of error is introduced into these data sets when they are interpolated across landscapes. The scenarios used for projections into the future are not meant to approximate future realities, which are unknowable. Rather, they are meant to represent plausible possible futures to facilitate discussion and planning (IPCC 2007). Moreover, the several climate models used in these projections differ in assumptions and outputs (IPCC 2007). We elected to average among scenarios and models to represent a midrange of their projections. While future vulnerability assessments of PAs should attempt to quantify and minimize uncertainty, resource management decisions will have to be made in the face of this uncertainty.

Management philosophy and approach

PA managers face a difficult challenge in selecting approaches that are both compatible with agency goals and likely to be effective under global change. The guiding policy for the newly formed NPS was to maintain parks in an "absolutely unimpaired form" and "faithfully preserve them for posterity" (National Park Service 2012:4). Consequently, many NPS managers operate under a philosophy of Ecological Process Management or Historic Range of Variation (HRV), wherein the goal is to maintain landscape patterns and ecological processes within the range of variation in place prior to Euro-American influence (Boyce 1998, Keane et al. 2009). Active management is used only where needed to restore lost ecological patterns or functions. Reintroducing an extirpated keystone predator would be an example of this type of management philosophy as was done with the reintroduction of the gray wolf (*Canus lupus*) into Yellowstone National Park.

An alternative view is that ecosystems have been or will be so altered by humans that current and future conditions will have little resemblance to "natural" historical conditions (Hobbs et al. 2010). Consequently, the goal of management should be to promote ecological integrity and resilience under future changing conditions (Colwell et al. 2012). This approach advocates designing future ecosystems based on ecological principles and using active management such as translocation of species to achieve ecological objectives.

Our results suggest that vulnerability assessment provides a foundation for tailoring management to individual PAs by quantifying magnitude of change in an ecosystem through time (Fig. 10). In systems undergoing little change, historic, current, and future ecosystem conditions may substantially overlap. In this case, adaptation strategies may not be needed because of little change and HRV may be most appropriate. Management to keep the system within the historic range is feasible and desirable. An example may be Olympic National Park on the Pacific Coast of the Northwest United States. Past and projected future climate change in this PACE is low to moderate due to the moderating influence of the Pacific Ocean. The large and long-lived rain forest trees species that dominate this PACE have high adaptive capacity and resiliency to shifts in climate and disturbance. Managers in this region are more focused on maintaining beneficial natural disturbance regimes within HRV than they are about the effects of climate change.

In fast changing ecosystems, current conditions may differ entirely from historic conditions and be on a trajectory to entirely new ecosystem states in the future (Fig. 10). Active management to create desired future conditions (DFC) may be the only viable management approach in this case. While the need for management may be high to avoid ecological degradation, the feasibility of such strategies may be low, the costs high, and the risk of unintended consequence high. In the Santa Monica Mountains PACE near Los Angeles, management to retain pre-EuroAmerican disturbance or ecological conditions will be either futile ecologically and/or socially unacceptable. Instead, managers there must decide what ecological conditions and ecosystem services they wish to achieve under future global change and develop active management strategies to produce and maintain these conditions and services.

Perhaps the most typical situation for parks in the United States is one where past, present, and future ecosystems overlap moderately (Fig. 10). In this case, analogs for ecosystems under future climate may already exist in particular biophysical settings. The key management goal in such landscapes may be to maintain/restore the mechanisms that promote resilience under changing conditions (Moritz et al. 2011). Such strategies are under development in and around the Lake Roosevelt PACE in the eastern Washington Cascades where topographic complexity and moderate climate change result in ecosystems in small watersheds lying along a gradient from historic to projected future conditions.

Policy implications

The results of this study support Colwell et al.'s. (2012) recommendation that the NPS steward its resources for continuous change that is not yet fully understood, to preserve ecological integrity. While some of the parks we studied experienced relatively little change in exposure to land use, climate and exotic species, other PACEs have undergone dramatic change in exposure, with projected increases in rates of change and subsequent ecological consequences. Two major policy implications emerge from this work: (1) PA managers can best develop PA-specific adaptation and management strategies to maintain desired ecological conditions by conducting vulnerability assessments across networks of PAs, and (2) adaptation planning and management of PAs under global change can best be done if explicit ecological goals are specified by guiding policy documents.

In the case of the NPS, policies and programs are largely in place that could facilitate vulnerability assessments. The NPS Inventory and Monitoring program (Fancy et al. 2009) was created to build regional networks of parks in support of monitoring natural resources consistently. More recently, the NPS Climate Change Strategy (National Park Service 2010) identified the need to conduct climate vulnerability assessments across all NPS units. A detailed framework for conducting such assessments (based on Glick et al. 2011) is under development with the NPS Intermountain Region (Whittington et al. 2013). While all three components of vulnerability pose scientific and logistic challenges, some components can be evaluated using existing data. Past exposure to land use, climate, invasive species, and possibly pollution can be reconstructed from historical data sources (this paper). Forecasts of potential future exposure are increasingly available from downscaled climate and land use models. Climate and land use observations and projections can be used to compare recent change with past and possible future change. The components of vulnerability that involve ecological response to exposure (sensitivity and adaptive capacity) often can only be quantified through scientific study that can be expensive and difficult. Fortunately, this science is increasingly available, including data on forest mortality and projections of vegetation response to climate change (e.g., Rehfeldt et al. 2012).

The NPS Inventory and Monitoring Program can greatly inform vulnerability assessment. The program currently monitors climate, human drivers, and conservation context through NPScape (Monahan et al. 2012), which utilizes many of the data sets analyzed here. Existing Inventory and Monitoring Program products could be further integrated with results generated by other divisions in the NPS to include additional PACE-level analyses of exposure, sensitivity, and adaptive capacity that are the basis for understanding vulnerability. Analysis and reporting of data on vulnerability across the network of parks would identify those units with similar threats, opportunities, and management solutions. The knowledge derived from these assessments would provide managers of individual parks with information that is vital to crafting management strategies for their parks in the context of other parks with similar vulnerabilities.

Vulnerability assessments are best done in the context of explicit conservation goals that can be used as a benchmark for comparison to current conditions and to guide management actions. Parks Canada (Parks Canada Agency 2008), for example, implemented a nationwide program for setting goals for ecological integrity across their park units and reporting at five-year intervals the condition of the park units relative to these goals. An independent scientific review of the U.S. NPS recommended a similar program be developed in order to better achieve the overarching goal of stewarding NPS resources to preserve them unimpaired under future change (Colwell et al. 2012). In accordance with this recommendation, the NPS has initiated *State of the Parks* reports that evaluate condition and trends in park resources relative to "reference conditions" (*available online*).[7] How best to define these reference conditions in terms of ecological integrity, execute these *State of the Parks* reports in the context of vulnerability assessments, and institutionalize the reports and assessments across the NPS system remains under discussion.

NPS and other PA management agencies face very real challenges with very limited resources (National Park Service 2010). Results of this study can facilitate global change adaptation by PA managers by (1) identifying trends in climate, land use, and other stressors already affecting their unit; (2) describing projected trends in the magnitude of stressors into the future; and (3) identifying other units facing similar challenges. The complexity and scale of global change requires collaborations unlike those typical of the past, and studies like this are necessary to promote partnerships and prepare PA managers to be effective stewards into the future.

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