Wilderness areas in a changing landscape: changes in land use, land cover, and climate

JoceLy n L. AyCrigg,1,4 T. RyAn McCarley,1 R. TrAvis Belote,2,3 and SeBasTian MaTrinuzzi3

1Department of Fish and Wildlife Sciences, College of Natural Resources, University of Idaho, Moscow, Idaho 83844 USA
2The Wilderness Society, Bozeman, Montana 59715 USA
3SILVIS Laboratory, Department of Forest and Wildlife Ecology, University of Wisconsin, Madison, Wisconsin 53706 USA

Citation: Aycrigg, J. L., T. R. Mccarley, R. T. Belote, and S. Martinuzzi. 2022. Wilderness areas in a changing landscape: changes in land use, land cover, and climate. Ecological Applications 32(1):e02471. 10.1002/eap.2471

Abstract. Wilderness areas are not immune to changes in land use, land cover, and/or climate. Future changes will intensify the balancing act of maintaining ecological conditions and untrammeled character within wilderness areas. We assessed the quantitative and spatial changes in land use, land cover, and climate predicted to occur in and around wilderness areas by (1) quantifying projected changes in land use and land cover around wilderness areas; (2) evaluating if public lands surrounding wilderness areas can buffer future land-use change; (3) quantifying future climate conditions in and around wilderness areas; and (4) identifying wilderness areas expected to experience the most change in land use, land cover, and climate.

We used projections of land use (four variables), land cover (five variables), and climate (nine variables) to assess changes for 707 wilderness areas in the contiguous United States by mid-21st century under two scenarios (medium-low and high). We ranked all wilderness areas relative to each other by summing and ranking decile values for each land use, land cover, and climate variable and calculating a multivariate metric of future change. All wilderness areas were projected to experience some level of change by mid-century. The greatest land-use changes were associated with increases in agriculture, clear cutting, and developed land, while the greatest land cover changes were observed for grassland, forest, and shrubland. In 51.6% and 73.8% of wilderness areas, core area of natural vegetation surrounding wilderness was projected to decrease for the medium-low and high scenarios, respectfully. Presence of public land did not mitigate the influence of land-use change around wilderness areas. Geographically, projected changes occurred throughout the contiguous U.S., with areas in the northeast and upper Midwest projected to have the greatest land-use and climate change and the southwestern U.S. projected to undergo the greatest land cover and climate change. Our results provide insights into potential future threats to wilderness areas and the challenges associated with wilderness stewardship and climate adaptation. Despite the high degree of protection and remoteness of wilderness areas, effective management and preservation of these lands must consider future changes in land use, land cover, and climate.

Key words: climate change; land cover; land use; landscape; National Wilderness Preservation System; protected areas; wilderness.

INTRODUCTION

Worldwide wild lands that are large in size and unfragmented have gained importance for conservation due to increases in anthropogenic land uses and impacts related to climate change (Watson et al. 2016, Hoffmann et al. 2019, Hoffmann and Beierkuhnlein 2020). Globally, these wild lands provide the last strongholds of intact ecosystems, reduce extinction risk for terrestrial biodiversity, store and sequester carbon, and reduce risks related to extreme climate events (e.g., floods, drought; Martin and Watson 2016, Watson et al. 2018, Di Marco et al. 2019). Approximately 10% of the area of these wild lands worldwide has been lost since the early 1990s (Watson et al. 2016). Furthermore, recent studies have shown that anthropogenic land uses have reduced biodiversity, altered ecological processes, and degraded ecosystem services in many places around the world, including wild lands (Foley et al. 2005, Newbold et al. 2016, Jones et al. 2018). In response, the Convention on Biological Diversity has proposed to reduce threats to biodiversity by addressing land-use change through retaining wild lands worldwide and restoring degraded natural ecosystems (Convention on Biological Diversity 2020). Change is inherent in all ecosystems, but knowing its source, magnitude, pace, and impacts is vital to
maintaining biodiversity, ecological processes, and ecosystem services. Even the National Wilderness Preservation System (NWPS) in the United States, which is comprised of >700 wilderness areas and has been designated a high level of protection through The Wilderness Act is not immune to changes in surrounding land use, land cover, and climate (wilderness.net, U.S. Public Law 88-577, Hansen and Defries 2007, Martiniuzzi et al. 2015, Newbold et al. 2016). The capacity of ecosystems to be resilient to change diminishes as the intensity and extent of change increases (Foley et al. 2005). Understanding changes in anthropogenic land use, natural land cover, and climate are necessary to maintaining wilderness areas as the cornerstone of the U.S. protected area network.

Human-driven changes degrade ecological conditions and processes surrounding wilderness areas, and are projected to intensify and expand (Foley et al. 2005, Hansen and Defries 2007, Radeloff et al. 2012). Changes in anthropogenic land use vs. changes in land cover are different and distinguishing between these two types of change is important for addressing their impacts. Anthropogenic land use (from this point forwards land-use) change is the total arrangements, activities, and inputs that humans undertake (e.g., development, mining, timber harvesting; Turner and Meyer 1994, IPCC 2000, Lambin et al. 2003). Natural land cover (from this point forwards land cover) change is change in the observed physical and biological terrestrial cover of the Earth as vegetation (e.g., grasslands, forests; Turner and Meyer 1994, IPCC 2000, Lambin et al. 2003). Urban development, mining, and agricultural activities, such as livestock grazing and farming, are examples of land-use change that transform existing land cover directly and indirectly by limiting ecological connections to surrounding lands (Theobald 2013, Venter et al. 2016). Given the high level of protection for wilderness areas, land-use activities indirectly affect wilderness by altering ecological processes, such as species migrations and the flow of nutrients, through alteration and degradation of the land cover surrounding wilderness areas (Hansen and Defries 2007, Dawson and Hendee 2009).

Wilderness areas provide core habitat (i.e., contiguous interior habitat) for maintaining biodiversity, intact ecological processes (e.g., flow of nutrients), and species population viability (Gaston et al. 2008, Belote et al. 2016, Baldwin and Beazley 2019). Maintaining core habitat, not only within wilderness areas, but also around wilderness areas results in less habitat fragmentation and isolation, which are major drivers of loss of habitat and biodiversity (Brooks et al. 2002, Mantyka-Pringle et al. 2012). Additionally, core habitat surrounding wilderness areas provides a buffer to the influence of land use and land cover change occurring outside wilderness areas and aids in conserving biodiversity (Belote and Wilson 2020).

Metrics used to measure the influence of climate change are categorized by the magnitude and timing of change as well as the availability and position of current conditions in the future (Garcia et al. 2014). Extreme weather events, such as drought, heat, cold, and precipitation, as well as the potential of wildfire weather gauge the magnitude of climate change, while seasonal differences in these extreme weather events address the timing of climate change (IPCC 2012, Garcia et al. 2014, Martiniuzzi et al. 2019, Zhang et al. 2019). Climate velocity and dissimilarity evaluate the future availability and position of current climate conditions (Garcia et al. 2014, Carroll et al. 2015). Knowing how wilderness areas will be influenced by changes in magnitude, timing, and availability and position of current climate conditions will allow wilderness managers to prioritize current management strategies and adaptation. Furthermore, climate-informed conservation strategies can be improved by identifying wilderness areas projected to undergo the most and least future changes due to climate (Belote et al. 2017a, b, 2018).

Previous studies that evaluated the effects of future global change on protected area networks typically considered land use, land cover, or climate change, separately, but not combined (Foley et al. 2005, Radeloff et al. 2012). Furthermore, these studies were based on a limited number of variables, such as change in development, agriculture, and logging for land use, urban expansion for land cover, and change in temperature and precipitation for climate (Radeloff et al. 2010, Seto et al. 2012, Wilson et al. 2015, Elsen et al. 2020). Few studies on land use, land cover, or climate change have focused on wilderness areas, which challenges wilderness managers to respond to this combination of stressors simultaneously. Therefore, an understanding of future changes in land use, land cover, and climate conditions affecting individual wilderness areas and across the NWPS is needed. We know that land use and land cover (LULC) change can intensify and accelerate climate change, but do not know the influence on wilderness areas or if wilderness areas can respond and adapt (Martin and Watson 2016).

For wilderness areas and wilderness managers, there is a balancing act between maintaining ecological conditions and untrammeled character as described in The Wilderness Act (U.S. Public Law 88-577, Hobbs et al. 2010, Aplet and Mckinley 2017). If LULC change in areas surrounding wilderness indirectly affects the ecological conditions within wilderness areas, should ecological conditions be actively restored, requiring management intervention, or should untrammeled character be upheld? Maintaining the untrammeled character of wilderness could lead to species loss, such as the keystone species whitebark pine (Pinus albicaulis, Holsinger et al. 2019). Alternatively, restoration of wilderness conditions through active management could address such species loss and maintain ecological conditions, but at the potential loss of untrammeled character (Hobbs et al. 2010, Appel 2015, Aplet and Mckinley 2017). Wilderness managers are currently grappling with
this balancing act and future changes in and around wilderness areas will heighten the importance of this balancing act.

Our goal was to assess the quantitative and spatial context of future changes in LULC and climate that are predicted to occur in and around wilderness areas. Our objectives were to (1) quantify future changes in LULC surrounding wilderness areas; (2) evaluate the extent to which public lands surrounding wilderness areas can act as a buffer to future land-use change; (3) quantify future changes in climate conditions in and around wilderness areas; and (4) identify those wilderness areas expected to undergo the most and least changes in LULC and climate. To address these objectives, we quantified changes between early to mid-century (~2050) and evaluated both the direction (i.e., positive or negative) and departure from current conditions to identify wilderness areas most influenced by changes in LULC and climate. We ranked wilderness areas using a multivariate metric of future change, which represented the influence of each LULC or climate change variable as well as groups of variables (e.g., all land-use variables) on wilderness areas. Using this metric, we ranked each wilderness area relative to all other wilderness areas within the NWPS. We provide wilderness managers with a perspective across the NWPS and a broader context for each individual wilderness area.

**Methods**

Data description and initial data processing

Wilderness areas and surrounding buffers.—We obtained boundaries for all wilderness areas within NWPS for the contiguous U.S. (CONUS) from Wilderness Connect (https://www.wilderness.net/). This included all wilderness areas managed by federal agencies (i.e., U.S. Fish and Wildlife Service, USDA Forest Service, Bureau of Land Management, and National Park Service), which totaled 714 wilderness areas. Although some wilderness areas are split by managing agency, we grouped wilderness areas by name for our analysis. Of the 714 wilderness areas within the contiguous U.S., seven were excluded from our analysis because of the lack of data (i.e., some spatial datasets did not cover all wilderness areas), which left us 707 wilderness areas for our analysis.

We quantified LULC change within a 10- and 20-km buffer surrounding wilderness areas. The results were comparable between the two buffer distances; therefore, we only summarized the results for the 10-km buffer distance. Furthermore, we chose a 10-km buffer to be consistent with previous studies that evaluated LULC change near protected areas (Radeloff et al. 2010, Martinuzzi et al. 2015). For climate variables, we summarized change in the wilderness areas plus the 10-km buffer to capture changes in surrounding land and to attribute small wilderness areas with coarse resolution climate data (i.e., 12-km data).

Future projections of LULC change.—We explored several datasets of future projections of LULC change for CONUS. However, because we wanted to compare LULC change with climate change, we chose the forecasting scenarios of land-use change projections (FORE-SCE) by Sohl et al. (2007, 2014), which were the only LULC dataset developed under socio-economic scenarios that approximated our emissions scenarios for climate change. These included the medium-low (a B1 world or RCP 4.5) and high (an A2 world or RCP 8.5; van Vuuren and Carter 2014) scenarios. We chose the term “scenario” for our future projections because we used a crosswalk developed by van Vuuren and Carter (2014) of the socio-economic scenarios with the emissions scenarios. The medium-low scenario focuses on environmental sustainability, which is characterized by low population growth, rapid technological innovation, a balanced energy strategy that includes renewable energy, and globalization (Sohl et al. 2014, 2016, van Vuuren and Carter 2014). The high scenario focuses on economic growth, which is characterized by high population growth, slow technological innovation, a regional energy strategy with a heavy reliance on fossil fuels, and regional growth rather than globalization (Sohl et al. 2014, 2016, van Vuuren and Carter 2014). The LULC projections of Sohl et al. (2007, 2014) focused on socio-economic scenario-based projections to provide the annual demand for LULC that we used for assessing LULC change in wilderness areas.

Sohl et al. (2014) created annual LULC 250-meter resolution maps for 2005–2100, although we used data for 2005 and 2050. The modeling output from FORE-SCE provided the 2050 projected values for all our LULC variables. Sohl et al.’s (2014) initial landscape conditions were set by the 1992 National Land Cover Database (NLCD), and 2005 was set as the baseline year for the projected period. The Protected Areas Database of the U.S. (PAD-US) was used to limit change within protected areas occurring within the 10-km buffer and wilderness areas (USGS 2016). More specifically, for the medium-low scenario, which focused on environmental sustainability, change was limited to lands with no protection, but for the high scenario, which focused on economic growth, change was limited to lands allowing multiple use or with no protection. Wilderness areas were protected from all LULC change other than natural vegetation succession. See Sohl et al. (2007, 2014) for more details.

We separated the LULC variables from Sohl et al. (2014) into two groups (i.e., land use and land cover) Our four land-use variables included developed, mining, clear cutting, and agriculture (Appendix S1: Table S1). Clear cutting is classified as a forest disturbance detected by the vegetation change tracker, which uses a time-series stack of Landsat images to detect forest disturbance (Huang et al. 2010). Clear cutting on private and public land was combined into a single variable, while...
cropland and hay/pasture were combined into an agricultural variable. Our five land cover variables included barren, forest, grassland, shrubland, and wetland (Appendix S1: Table S1). Deciduous, evergreen, and mixed forest were combined into a single forest class while wetland included a combination of herbaceous and woody wetland.

We used model output from Sohl et al. (2014) to assess LULC changes from 2005 to 2050. We calculated percent change (increase or decrease) in area during 2005–2050 for each LULC variable within the 10-km buffer surrounding wilderness areas (Appendix S1: Table S1). We designated an increase in percent change as >1% and a decrease in percent change as <−1%. Future changes in LULC variables ranged from negative to positive values because departure from current conditions could either indicate an increase or decrease in percent change of area. The percent change in area for each LULC variable for each wilderness area buffer was used to determine decile values and a multivariate metric of change (see Ranking of wilderness areas below for more details).

Changes in LULC alter existing vegetation and reduce the amount of natural vegetation that is core habitat. To assess the amount of core habitat surrounding each wilderness area, we used our selected land cover classes based on Sohl et al. (2014; i.e., barren, forest, grassland, shrubland, and wetland) plus perennial ice/snow to represent natural vegetation. We classified core pixels using Morphological Spatial Pattern Analysis (MSPA; Soille and Vogt 2009) in the Graphical User Interface for the Description of Image Objects and their Shapes (Appendix S2: Fig. S1, GuidosToolbox; Vogt and Ritters 2017). We evaluated multiple distances (250 and 1,000 m) from the edge of natural vegetation patches toward the center and determined that an edge distance of 1,000 m best represented the changes occurring around wilderness areas with respect to core habitat. This edge distance struck a balance between capturing changes in core habitat for small patches and large patches of natural vegetation (Appendix S2: Fig. S1).

Using the percent area of natural vegetation for 2006 and 2050, we calculated the difference in percent area to determine the amount of change estimated to occur in natural vegetation by 2050 under two scenarios. The difference in percent area of core habitat included both negative (i.e., reduction in amount of natural vegetation) and positive (i.e., increase in amount of natural vegetation) values. A decile value and a multivariate metric of future change were calculated for each wilderness area based on the change in percent area of core habitat (see Ranking of wilderness areas below for more details).

Public lands data.—To evaluate the extent to which public lands surrounding wilderness areas can act as a buffer to future land-use change, we used the geodatabase PAD-US (version 1.4), which is an inventory of public land and private protected areas, to assess the amount of public land surrounding wilderness areas (USGS 2016). Public lands included land managed by federal and state agencies (e.g., USDA Forest Service and state natural resource agencies).

To evaluate if the amount of public land surrounding wilderness areas provided a buffer from land-use change, we used PAD-US to calculate the total area and percent area of public land within each 10-km buffer surrounding wilderness areas (USGS 2016). For each wilderness area, we calculated the percent public land in relation to projected percent increase (i.e., change >0) of land-use classes (i.e., agriculture, clear cutting, developed, and mining) from 2006 to 2050 using the medium-low and high scenarios. For instance, for each wilderness area, we plotted the percent increase of developed land within the 10-km buffer with the percent public lands under both medium-low and high scenarios. We used a regression analysis and calculated coefficients of determination (i.e., $R^2$ values). If public lands are buffering wilderness areas from surrounding land-use change then we would expect less land-use change as measured by projected percent change with higher percentages of public land within the 10-km buffer.

Future projections of climate change.—We focused on nine climate related variables for the medium-low and high scenarios. We categorized our climate variables by magnitude and timing of climate change as well as by the availability and position of future conditions (Garcia et al. 2014) under each scenario.

For the magnitude and timing of climate change, we summarized the change in frequency of extreme weather variables during 2006 to 2050 using our two scenarios over four seasons (spring, summer, fall, and winter). Extreme weather variables were defined as those with a 20-yr return interval observed in the historic daily record (1950–2005; IPCC 2012). We included four seasons to represent the seasonal variation in the timing of climate changes. We used five extreme weather variables; heat, cold, drought, precipitation, and false springs, which is the number of days between first bloom and last hard freeze (a daily minimum temperature $<−2.2\, ^\circ C$; Marino et al. 2011, Martinuzzi et al. 2016, 2019: Appendix S1: Table S2).

Four of our extreme weather variables (i.e., cold, drought, heat, and precipitation) were derived from daily Standardized Precipitation Evapotranspiration Index (SPEI) and Standardized Temperature Index (STI) records using 19 General Circulation Models (GCMs) downscaled to 12 km as described in Martinuzzi et al. (2016). We calculated the future return interval during 2006–2050 for extreme weather events, including drought, heat, precipitation, and cold for each of 19 GCMs by counting the number of SPEI or STI values projected to exceed the historic 20-yr return interval threshold (Martinuzzi et al. 2016). For each wilderness area, we calculated the difference in return interval using the median value for 19 GCMs in the historic and projected periods. We calculated these extreme weather...

---

**Ranking of wilderness areas**

The percent change in area for each LULC variable within the 10-km buffer surrounding wilderness areas (Appendix S1: Table S1). We designated an increase in percent change as >1% and a decrease in percent change as <−1%. Future changes in LULC variables ranged from negative to positive values because departure from current conditions could either indicate an increase or decrease in percent change of area. The percent change in area for each LULC variable for each wilderness area buffer was used to determine decile values and a multivariate metric of change (see Ranking of wilderness areas below for more details).

Changes in LULC alter existing vegetation and reduce the amount of natural vegetation that is core habitat. To assess the amount of core habitat surrounding each wilderness area, we used our selected land cover classes based on Sohl et al. (2014; i.e., barren, forest, grassland, shrubland, and wetland) plus perennial ice/snow to represent natural vegetation. We classified core pixels using Morphological Spatial Pattern Analysis (MSPA; Soille and Vogt 2009) in the Graphical User Interface for the Description of Image Objects and their Shapes (Appendix S2: Fig. S1, GuidosToolbox; Vogt and Ritters 2017). We evaluated multiple distances (250 and 1,000 m) from the edge of natural vegetation patches toward the center and determined that an edge distance of 1,000 m best represented the changes occurring around wilderness areas with respect to core habitat. This edge distance struck a balance between capturing changes in core habitat for small patches and large patches of natural vegetation (Appendix S2: Fig. S1).

Using the percent area of natural vegetation for 2006 and 2050, we calculated the difference in percent area to determine the amount of change estimated to occur in natural vegetation by 2050 under two scenarios. The difference in percent area of core habitat included both negative (i.e., reduction in amount of natural vegetation) and positive (i.e., increase in amount of natural vegetation) values. A decile value and a multivariate metric of future change were calculated for each wilderness area based on the change in percent area of core habitat (see Ranking of wilderness areas below for more details).
values for four seasons (i.e., spring, summer, fall, and winter). However, for our extreme cold variable, we only included winter, and for false springs events we only included spring because spring is the only season in which these events occur (Appendix S1: Table S2).

We used a dataset created by Martinuzzi et al. (2016) based on data generated by Allstadt et al. (2015) to determine the frequency of false springs, which is expressed as a return interval in years. False springs, as described by Martinuzzi et al. (2016), captured the mean annual probability of false springs for each pixel during the historic record and during 2006–2050 in which reproductive effort by plants may have not been successful because of hard freezes occurring after blooming. The return interval of false springs, which is the average time between false spring occurrences, was calculated by dividing one by the annual probability (Martinuzzi et al. 2016, 2019).

Another variable we used to assess the magnitude of climate change was the potential for wildfire weather using the Keetch-Byram Drought Index (KBDI) for 12 km pixels based on daily precipitation, temperature, length of day and weather conditions on the previous day (Appendix S1: Table S2; Keetch and Byram 1968). The KBDI measures wildfire potential in the contiguous U.S. by estimating daily soil moisture without making assumptions about vegetation growth (Keetch and Byram 1968, Martinuzzi et al. 2019).

The 95th percentile of the KBDI for 1950–2005 was used to determine an historical extreme value, which was compared with projected estimates to determine the average number of days KBDI would be above the historical threshold in 2050 (Martinuzzi et al. 2019). Increases in the number of days above the historical KBDI 95th percentile relative to the historic conditions indicated an increase in wildfire potential.

To assess the availability and position of current climate conditions in the future, we used climate velocity (i.e., forwards, and backwards) and climate dissimilarity (Appendix S1: Table S2). Forwards and backwards climate velocities were obtained from Carroll et al. (2015) and represent expected geographic displacement of multivariate climate analogs (i.e., species a species would need to move to track changes in climate) for 2055 (i.e., mid-century). Specifically, forward velocity represents the distance between current climate to future conditions (i.e., where climate analogs are expected to move; Carroll et al. 2015). Backward climate velocity represents the distance from where future climate analogs are expected to arrive (i.e., where climate analogs are coming from; Carroll et al. 2015). Climate dissimilarity is an index that summarizes the magnitude of multivariate climate change projected for each wilderness area plus its 10-km buffer centered on 2055 (Belote et al. 2018). The projected climate dissimilarity was calculated as the Euclidean distance between current (i.e., average climate between 1981 and 2010) and future (i.e., 2014–2070, centered on 2055) climate (see Belote et al. 2018 for details).

Climate velocity, both forwards and backwards, was assessed by overlaying climate velocity data from Carroll et al. (2015) onto our wilderness areas plus the 10-km buffer areas to calculate a mean velocity for each wilderness area. We used climate dissimilarity to assess the position of current climate conditions in the future. For climate dissimilarity, we overlaid data from Belote et al. (2018) onto our wilderness areas plus 10-km buffer areas to calculate a mean dissimilarly index value for each wilderness area.

All our climate change variables, which included change in return interval in years for our extreme weather data, change in days above 95th percentile for KBDI, kilometers per year for climate velocity and an index value for climate dissimilarity, were determined for each wilderness area and the surrounding 10-km buffer. Future changes in climate variables ranged from negative to positive values because departure from current conditions could either indicate an increase or decrease in percent change of area. A decile value and multivariate metric of future change were determined for each of these climate change variables for ranking of wilderness areas (see Ranking of wilderness areas below for more details).

Data analysis

Ranking of wilderness areas.—To rank wilderness areas, we combined LULC and climate change variables into a multivariate metric of future change that represented the influence of LULC and climate change on wilderness areas and their surrounding 10-km buffer. Initially, we examined correlations between each combination of land use, land cover, and climate change variables. If two variables had a correlation value of >0.70, one of the variables was dropped from further analysis.

To composite multiple variables into the multivariate metric of future change, we first assigned a decile value (i.e., an integer from 1 to 10) to each LULC and each climate change variable based on the distribution of values across all wilderness areas and across both scenarios. Each decile value represented a bin containing 10% of the distribution of values for each LULC or climate change variable. For example, the decile values were designated for core habitat of natural vegetation by grouping the distribution of values into 10 bins with 10% of the distribution of values in each bin. This put all wilderness areas with their associated variables on the same scale relative to all other wilderness areas and comparable between scenarios. Higher decile values indicated increasing departure from historic conditions. We summed the decile values for each wilderness area to obtain our multivariate metric of future change with values ranging from 0 for the lowest summed values to 100 for the highest summed values.

Our multivariate metric of future change was calculated across four separate categories: (1) land-use variables (i.e., developed, mining, clear cutting, and
agriculture), (2) land cover variables (i.e., barren, grassland, shrubland, forest, wetland, and core habitat for natural vegetation), (3) the aggregate of LULC variables, and (4) all climate change variables. Each wilderness area was ranked relative to all other wilderness areas using our multivariate metric of future change, which was a value from 0 to 100 for each of the four categories.

To identify wilderness areas expected to undergo the most and least changes in land use, land cover, and climate, we evaluated four comparisons among all wilderness areas using our composite multivariate metric of future change: (1) land use vs. land cover, (2) land use vs. climate, (3) land cover vs. climate, and (4) LULC vs. climate. To assess the quantitative differences between wilderness areas for each of the four comparisons, we plotted all wilderness areas by their multivariate metric of future change in a scatter plot. To spatially assess differences between wilderness areas, we mapped all wilderness areas by their multivariate metric of future change for each of the four comparisons. We created a scatter plot and a map for each of our two scenarios for each of our four comparisons.

**RESULTS**

**Wilderness areas and surrounding buffer**

We included 707 wilderness areas in our analysis around which we created a 10-km buffer (Fig. 1). These 707 wilderness areas total 21,911,024 ha in the contiguous U.S. with Pelican Island Wilderness in Florida being the smallest (~2 ha) and Death Valley Wilderness in California and Nevada being the largest (~1.2 million ha). Most wilderness areas occur west of the Mississippi River with California having the highest number of wilderness areas and total area of wilderness. Wilderness areas plus their 10-km buffer include 11.8% of the total area of CONUS.

**Future projections of LULC change**

Using the projected values from the Sohl et al. (2014) model wilderness areas were projected to depart from current conditions, which was most evident based on increases in developed and agriculture (Fig. 2). For example, 11.0% of wilderness areas (n = 77) were projected to see an increase in developed areas for the high scenario. Clear cutting increased more under the high scenario (24.3%) compared with the medium-low scenario (10%). But there were also indications that clear cutting was projected to decrease by mid-century around some wilderness areas (Fig. 2). However, on average, 91% of wilderness areas in the medium-low scenario and 85% in the high scenario were projected to have little to no change in surrounding land use (i.e., developed, mining, clear cutting, and agriculture, Fig. 2).

Forest, grassland, and shrubland showed the greatest departure from current conditions among land cover variables (Fig. 2). Differences between the medium-low and high scenarios were more pronounced for both forest and wetland land cover variables (Fig. 2). Like land use, most wilderness areas showed little to no change (medium-high scenario mean: 94%; high scenario mean 90%) in projected values from Sohl et al. (2014) model output for surrounding land cover (i.e., barren, grassland, forest, shrubland, and wetland).

For core habitat comprised of natural vegetation, our results indicated that 51.6% and 73.8% of wilderness areas will undergo a loss (i.e., decrease in percent area) of core habitat area by mid-century for the medium-low and high scenarios, respectfully (Fig. 2).

**Public land surrounding wilderness areas**

The percent of public land area within the 10-km buffer around wilderness areas did not appear to buffer land use change (Fig. 3). However, the percent change in developed areas and agricultural areas was negatively associated with percent of public land area within the 10-km buffer, which fits with our expectation of less land use change associated with higher percent area of public land.

**Future projections of climate change**

Using the change in return interval of extreme weather events (i.e., drought, heat, precipitation, cold, and false springs) to assess the magnitude of climate change and using seasonal (i.e., winter, summer, spring, and fall) values to evaluate the timing of climate change, we found substantial variability among wilderness areas both within and between seasons for extreme events (Fig. 4). For instance, extreme drought events were projected to become more frequent during summer compared with fall, spring and winter. Overall, extreme weather events were projected to change in frequency by mid-century in and around wilderness areas with the return interval of droughts and heat in all seasons projected to decrease (i.e., become more frequent and have a negative change in return interval). Precipitation in all seasons, except winter for the high scenario, was projected to increase (i.e., become less frequent and have a positive change in return interval) in and around wilderness areas (Fig. 4). The return interval for winter cold was projected to increase (i.e., become less frequent), which indicates fewer cold winters by mid-century (Fig. 4). The mid-century projection for the return interval of false springs indicated both an increase and decrease in frequency.

We projected that KBDI, which quantifies the potential frequency for wildfire weather, would be about 35–51 d above the historical threshold by 2050 with all wilderness areas indicating an increase in KBDI (Fig. 4). The high scenario predicted that 52.3% of wilderness areas would have a KBDI value >51, the median value, which means that 51 additional days of extreme fire weather by 2050 were projected compared with current conditions.
When assessing the availability and position of current climate conditions in the future, we found that median forward velocity was projected to be 1.3 km/yr (range = 0.1–14.3 km/yr) for the medium-low scenario and 2.1 km/yr (range = 0.2–22.7 km/yr) for the high scenario (Fig. 4). Backward velocity was projected to be 1.4 km/yr (range = 0.2–45.8 km/yr) for the medium-low scenario and 2.4 km/yr (range = 0.2–50.0 km/yr) for the high scenario. All wilderness areas were projected to have climate conditions different from their historical average climate conditions based on our projected climate dissimilarity index values for both scenarios (Fig. 4).

**Ranking of wilderness areas**

We designated a decile value for each LULC, and each climate change variable based on the distribution of values across all wilderness areas and we mapped each variable separately (Appendix S3: Figs. S1–S8). This information can be used to evaluate each LULC and climate variable and to determine which variable(s) will have the most influence in the future on each wilderness area (Data S1: Tables S1, S2; Data S2: Tables S1–S5). Below we describe overall patterns of decile values. Summed decile values were used to determine the multivariate metric of future change for ranking wilderness areas.

We identified wilderness areas projected to undergo the most and least change in the future relative to all other wilderness areas by ranking them using our multivariate metric of future change within four comparisons: land use vs. land cover, land use vs. climate, land cover vs. climate, and LULC vs. climate (Figs. 5, 6; Appendix S1: Table S1). These four comparisons indicated whether future changes in land use, land cover, and/or climate will have the most influence on each individual wilderness area. Standardizing the ranking of wilderness areas using our multivariate metric of future change put values for both scenarios on the same scale. Therefore, there were fewer wilderness areas ranked high in our comparisons of the medium-low scenario relative to the high scenario (Figs. 5, 6, Table 1).

**Decile values.**—Using the decile values, the patterns in projected changes varied among LULC variables with clear cutting, agriculture, grassland, and shrubland having more wilderness areas with the highest decile values for both scenarios compared with the other LULC variables (Appendix S3: Figs. S1, S2, Data S1: Tables S1, S2). High decile values of core areas of natural vegetation within the 10-km buffers occurred throughout the U.S. for both scenarios (Appendix S3: Fig. S3).

Among the climate change variables, the highest decile values occurred in the western U.S. during spring and...
summer for extreme drought, the northwestern U.S. during spring and southwestern U.S. during summer for extreme heat, the western U.S. during spring, summer, and fall for extreme precipitation, and in the western U.S. for winter cold and false springs (Appendix S3: Figs. S4–S7, Data S2: Tables S1–S4). KBDI decile values were highest in the upper Midwest and western U.S. (Appendix S3: Fig. S8, Data S2: Table S5). Climate velocity (forwards and backwards and climate dissimilarity decile values were high throughout the U.S. for both scenarios (Appendix S3: Fig. S8, Data S2: Table S5).

LULC change comparison.—Departure from current conditions relative to all wilderness areas represented by our multivariate metric of future change showed that LULC are changing around many wilderness areas (Fig. 5a, b). Changes in land cover corresponded to a change in land use because natural vegetation change (e.g., succession) is not captured in the Sohl et al. (2014) model. Therefore, any change in land cover corresponds to a change in land use. All wilderness areas with a high value for our multivariate metric of future change occur in the western U.S. (Table 1, Fig. 5a, b). The wilderness areas predicted to undergo the least amount of LULC change occurred throughout the U.S. for the medium-low scenario but occurred mostly in the western U.S. for the high scenario (Fig. 5a, b).

Land use vs. climate change comparison.—The comparison of projected changes in land use and climate indicated that under the medium-low scenario, some wilderness areas were ranked high (i.e., >67) using the multivariate metric of future change for land use and others ranked high for climate, but none ranked high for both (Fig. 5c, Table 1). Contrastingly, under the high scenario, there were 21 wilderness areas ranked high for both land use and climate change (Fig. 5d, Table 1). Geographically, the highest ranked (i.e., >67) wilderness areas occurred mostly in the upper midwest and northeastern U.S. for the high scenario, while the lowest ranked (i.e., <33) wilderness areas occurred mostly in Florida (Fig. 5d, Table 1).

Land cover vs. climate change comparison.—Patterns of projected change in land cover and climate under the medium-low scenario were like the patterns of projected change in land use vs. climate (Fig. 5). There were wilderness areas ranked high (i.e., >67) for one or the other (i.e., land cover or climate, separately), but no wilderness area ranked high for both (Fig. 5e, Table 1). However, 17 wilderness areas were ranked high for both land cover and climate change under the high scenario, with all occurring in western U.S. (Fig. 5f, Table 1). Under the high scenario, there were nine wilderness areas that ranked the lowest for land cover and climate,
all of which occurred in the southeastern U.S. (Fig. 5f, Table 1).

**LULC vs. climate change comparison.**—Evaluation of the comparison of future change for LULC vs. climate variables indicated that all wilderness areas are being influenced by both LULC and climate change under both scenarios (Fig. 6). Because we standardized our ranking across the medium-low and high scenarios, it is not surprising that under the high scenario more wilderness areas had a multivariate metric of future change value >33 for LULC and climate change than under the medium-low scenario (Fig. 6). Furthermore, under the high scenario, more wilderness areas had a multivariate metric of future change value >67 for climate change compared to LULC (Fig. 6b).

Geographically, most of the projected change in LULC occurred in the northwestern US (e.g., Washington and Oregon) and central Colorado, particularly under the high scenario (Fig. 6b). While the change in climate was mainly in the southwestern U.S. (Fig. 6b). Most wilderness areas fell within the middle portion of the scatter plots (green symbols in Fig. 6) indicating that relative to all other wilderness areas most were undergoing similar amounts of projected change in LULC and climate. Under the high scenario, 17 wilderness areas were projected to have the greatest change in departure and 13 (76%) were distributed in the western U.S. (i.e., Arizona, California, Colorado, Nevada, New Mexico, and Oregon) with the remaining four (24%) occurring in Illinois, Michigan, and Pennsylvania (Table 1). The five wilderness areas ranked the lowest for LULC and climate change under the high scenario occurred in Florida, Georgia, and Mississippi (Fig. 6).

**DISCUSSION**

Wilderness areas are not immune to change. As soon as mid-century (i.e., ~2050) all wilderness areas will be influenced to some extent by land use, land cover, and/or climate change. Most wilderness areas are projected to have a multivariate metric of future change value >33 for LULC (49.6% for medium-low and 76.9% for high scenarios) and climate change (72.7% for medium-low and 95.6% for high scenarios; Fig. 6; Data S3: Table S1). This projection was also true for our comparisons of land cover vs. climate and land use vs. climate (Fig. 5). Projected changes in climate and land cover as well as LULC and climate were greatest within wilderness areas in the southwestern U.S., while projected land-use and climate changes were greatest around wilderness areas in the upper midwest and northeastern U.S. (Figs. 5d, 6b, Table 1). These projected changes, are vital to understanding how wilderness areas can adapt and how wilderness managers can manage for adaptation, particularly for climate change adaptation. Even though LULC change in areas surrounding wilderness areas indirectly influences wilderness areas, climate change can drive both land-use and land cover change, which ultimately leads to change in the ecological processes upon which species and vegetation communities within wilderness areas depend (Blois et al. 2013, Peters et al. 2019).

**Future projections of LULC change**

The influence of LULC change cannot be understated because it is estimated that >77% of the global land...
surface is directly influenced by humans (Watson et al. 2018). Wilderness managers need information on how wilderness areas will be influenced by changes in LULC to guide strategies for mitigation and adaptation. By mid-century under the high scenario, which focuses on economic development and is widely considered the current path for climate projections (Walsh et al. 2014, Hayhoe et al. 2017), the greatest land-use change in areas surrounding wilderness areas are projected to be in agriculture and clear cutting across the U.S. (Fig. 2; Appendix S3: Fig. S1). These projected changes are likely to be driven by high population growth and economic conditions that influence commodity prices (Sohl et al. 2014, Stehfest et al. 2019). Developed areas will...
also increase, but geographically they are not as widespread and instead are limited to specific areas, such as near existing cities (Appendix S3: Fig. S1; Wilson et al. 2015). Similar changes were projected around National Wildlife Refuges by Hamilton et al. (2013) and Martinuzzi et al. (2015), which is interesting because National Wildlife Refuges are typically at lower elevations surrounded by private lands, while wilderness areas are located in higher elevations surrounded by public lands. Because each of these land-use variables (i.e., agriculture, clear cutting, and development) will have different influences on wilderness areas, wilderness managers need to be mindful of wilderness areas undergoing the greatest projected land-use change and specifically incorporate protections that mitigate development.

Changes in human population growth and advances in renewable energy development have led to projected changes in land cover variables (Sohl et al. 2014). We projected decreases in grasslands, forests, and shrublands under both the medium-low and high scenarios (Fig. 2; Appendix S3: Figs. S1, S2). Grasslands and shrublands were projected to have the greatest decline with most being converted to agriculture and development in California (Wilson et al. 2015) This follows with our projections of changes in grasslands and shrublands mainly occurring in the western U.S. (Appendix S3: Fig. S2). Contrastingly, changes in forests were projected throughout the U.S. (Appendix S3: Fig. S2), which is likely to be attributed to forests undergoing clear cutting because patterns of projected change in clear cutting are similar to those of forest land cover change (Appendix S3: Figs. S1, S2).

Our projections indicate that, within the 10-km buffer surrounding wilderness areas, more than half of all wilderness areas will have a loss of core habitat under the medium-low and high scenarios, respectively. Loss of core habitat increases habitat fragmentation and isolation of wilderness areas, which is a major driver of biodiversity loss (Goetz et al. 2009, Radeloff et al. 2010, Haddad et al. 2015, Belote et al. 2016). By losing adjacent core habitat, the effectiveness of wilderness areas to protect the species, ecological processes, and biotic interactions for which they were established will be compromised (Gimmi et al. 2011, Haddad et al. 2015). Furthermore, the loss of core habitat is a pattern observed globally that has led to focusing on connectivity between protected areas, particularly large protected areas with minimal human influence, such as wilderness areas (Haddad et al. 2015, Belote et al. 2016). Belote et al. (2016) found the potential for connecting large, protected areas was greatest in the western U.S. with less potential in the eastern U.S. However, our spatial projections indicated that the greatest loss of core habitat surrounding wilderness areas will occur throughout the U.S., with no particular region having the highest loss (i.e., there were high decile values distributed across
Therefore, even though the potential for connectivity between wilderness areas is greater in the western U.S., the need for connectivity is throughout the U.S. Furthermore, this indicates that the loss of core habitat is likely to be related to projected changes in land use, land cover, and/or climate around most wilderness areas (Seto et al. 2012, Haddad et al. 2015, Hoffmann et al. 2019). The spatial distribution and projected area of core habitat loss will require wilderness and federal agency managers to focus on minimizing fragmentation around wilderness areas rather than mitigating or adapting to this projected loss.

A review of multiple LULC models, including the FORE-SCE model, showed high variability in projected trends and spatial patterns between models (Sohl et al. 2016). This indicates that there is uncertainty in our future projections of LULC change. However, the variability between models was minimal for projections of change in developed areas, which could be related to development occurring in specific areas. Those wilderness areas with projected change in development in surrounding areas probably contain less uncertainty because of the consistent output between models. Our choice of the FORE-SCE model was based on its annual output and chosen scenarios because they best matched our climate data, but we encourage wilderness managers to compare our results with other LULC projections and with LULC data collected for individual wilderness areas.

Public land surrounding wilderness areas

Even though, on average across the NWPS, 68.3% of the 10-km buffer surrounding wilderness areas is public land, the presence of public land alone is a poor predictor of whether a wilderness area will be influenced by land-use change (Fig. 3). Martinuzzi et al. (2015) found that wilderness areas, in comparison with other protected areas (e.g., National Wildlife Refuges), are projected to undergo the least amount of land-use change but they still are projected to experience increased land-use change by 2051. Furthermore, Martinuzzi et al. (2015) projected the amount of land-use change due to urban expansion will increase 37–70% by 2051 around wilderness areas, which was comparable with their projections for urban expansion around all protected areas within CONUS (29–72%).
The low correlation we observed between amount of public land surrounding wilderness areas and land-use change could be attributed to the multiple ways in which public land is used (i.e., multiple use) and that, under the medium-low scenario, multiple-use lands were not included in the model projections (Fig. 3). Many public lands, such as lands managed by Bureau of Land Management (BLM), and USDA Forest Service (USDA-FS), are mandated for multiple use, which includes timber harvesting, mining, energy development, and livestock grazing in addition to conserving natural resources (Federal Land Policy and Management Act of 1976 [43 USC § 1701], Multiple-Use Sustained Yield Act of 1960 [16 USC § 528]). Of the 68.3% of public land surrounding wilderness areas, 27.3% is managed by BLM and 53.2% is managed by USDA-FS, which totals 80.5% under multiple-use management. Under projections for the medium-low scenario the public lands managed by BLM and USDA-FS remained static, which probably influenced our results because they make up most of the public land surrounding wilderness areas. These lands did not remain static under the high scenario projections because of the high scenario, which focused on economic growth, and which increased land-use change (i.e., clear cutting) on multiple-use lands.

Public land might serve as a buffer for land-use change for some wilderness areas. There were 195 (28%) wilderness areas with >90% of the 10-km buffer in public land and under the high scenario, all were projected to have <4% development by mid-century. These included Yosemite and Sequoia-Kings Canyon wilderness areas in California as well as Government Peak Wilderness in Nevada. Furthermore, there was a negative relationship between the percent change in developed and agricultural areas compared with percent area of public land within the 10-km buffer (Fig. 3).
conservation of species, habitat, and ecological processes more than resource extraction then they could act as a buffer to change in other land uses, such as development and agriculture (Aycrigg et al. 2013, Belote and Wilson 2020). But across the NWPS, we cannot presume that the influence of land-use change can be mitigated even when wilderness areas are surrounded by public land especially if policies change to allow increased resource extraction.

Future projections of climate change

Wilderness areas are less directly influenced by land use because of the protection provided by The Wilderness Act (U.S. Public Law 88-577), but all protected areas, including wilderness areas, are influenced by climate change. We chose to evaluate the change in frequency of extreme weather events because species and vegetation communities can be drastically altered by a single extreme event and a population’s ability to adapt to extreme events is limited (Parmesan et al. 2000, McKechnie and Wolf 2010, Martinuzzi et al. 2016). Our mid-century projections for both medium-low and high scenarios indicate changes in climate with an increase in magnitude (i.e., more frequent occurrences) and in seasonal timing (i.e., occurring across all seasons) within wilderness areas, such as extreme heat and extreme drought (Fig. 4). This pattern is consistent with climate projections worldwide (IPCC 2012). Projections of more frequent extreme heat and drought events are particularly concerning because of the direct links between extreme heat and bird mortality (McKechnie and Wolf 2010) and extreme drought and species composition, such as a drought in Bandelier Wilderness, New Mexico that led to a die-off of ponderosa pine (Pinus ponderosa) and an increase in pinon-juniper woodlands (Allen and Breshears 1998). Wilderness areas in southwestern U.S., such as the Bandelier Wilderness, are projected to undergo the greatest changes in magnitude for extreme drought (Appendix S3: Fig. S4).

As the magnitude and timing of extreme events are projected to change, the disturbance to species and ecosystems could lead to changes in species composition and the re-organization of entire ecosystems within wilderness areas (Parmesan et al. 2000, Pecl et al. 2017, Maxwell et al. 2019). Maxwell et al. (2019) found that changes in species composition, particularly for invertebrates, were commonly observed with extreme precipitation events. They also found that 38% of observational studies indicated no recovery of ecosystems from extreme precipitation events within about 2 yr (Maxwell et al. 2019). Even though our projections for the high scenario indicate that the magnitude of extreme precipitation events will become less frequent (i.e., the change in return interval is positive in all seasons except winter), especially in the southwestern U.S., the intensity and duration of these events could increase (Fig. 4; Appendix S3: Fig. S6).

The influence of extreme cold during winter on species populations and ecosystems will be a function of severity (i.e., magnitude) and frequency (Williams et al. 2015). Our projections indicate that extreme cold during winter will decrease to a return interval of about every 25 yr by mid-century within the western and northern regions of the U.S. (Fig. 4; Appendix S3: Fig. S7). Winter cold temperatures constrain the geographic distribution of many species and with fewer projected extreme cold events in the future, some species could expand their distribution (i.e., poleward or altitudinally upwards), such as the mountain pine beetle (Dendroctonus ponderosae; Williams et al. 2015, Buotte et al. 2017). The mountain pine beetle has contributed to the decline of the white-bark pine (Pinus albicaulis) in alpine areas of wilderness areas in the western U.S., which helped to necessitate protection under the Endangered Species Act (USFWS 2011, Buotte et al. 2017).

Of all the climate variables we assessed for changes in magnitude and timing, the occurrence of false springs could have the most direct impact on vegetation communities (Inouye 2000, 2008, Kral-O’Brien et al. 2019). False springs are the number of days between first bloom and last hard freeze. They represent years in which reproductive effort by plants may have not been successful because of blooming occurring before the last hard freeze (Marino et al. 2011, Martinuzzi et al. 2016, 2019). Under the high scenario, we projected that 18.2% of wilderness areas will have more frequent false springs in the southwestern U.S. and along the west coast. While 23.8% will have less frequent false springs (Fig. 4). However, most wilderness areas (56.4%) did not have a projected change in frequency of false springs under the high scenario. Even though, Peterson and Abatzoglou (2014) found a decrease in frequency of false springs across CONUS during 1920–2013, there are likely to be regional differences, as we projected, because of differences in latitude and elevation. Inouye (2008) and Augspurger (2013) observed increases in the frequency of false springs in high elevations of Colorado and temperate forests in Illinois.

Within wilderness areas projected to have more frequent false spring, the vegetation communities may shift toward plant species more tolerant of these conditions, which ultimately could alter the diversity and composition of existing vegetation communities (Inouye 2008, Pardee et al. 2018). Pardee et al. (2018) found that two out of three wildflower species were directly impacted by frost events during spring, while only one in three species exhibited negative indirect effects on plant reproduction. This variability in response to false springs could lead to changes in vegetation communities within wilderness areas projected to experience more frequent false springs by mid-century and could lead to changes in resource availability, species diversity, and species populations.

Our projected change in magnitude and timing for the potential frequency of wildfire weather by 2050 (i.e., change in KBDI) corresponds to the increase in
frequency of extreme heat and the decrease in frequency of extreme precipitation during summer and fall when many wildfires occur (Fig. 4). The combination of more frequent high heat and less precipitation would create drier conditions in many wilderness areas creating a higher potential for wildfires (IPCC 2012, Barbero et al. 2015). Abatzoglou and Williams (2016), using multiple metrics of fire aridity (i.e., high heat and less precipitation), including KBDI, showed an increase in fire aridity from 2000 to 2015 across western forests in the U.S. Martinuzzi et al. (2019) also found an increase in the days of extreme fire weather for USDA Forest Service ranger districts with 72% projected to see >60 additional days of extreme fire weather. Barbero et al. (2015) found that the highest probability of increased potential of very large fires (i.e., top 5–10% of largest fires) occurred in the western U.S., which is the region in which our projections indicated a higher potential frequency of wildfire (Appendix S3: Fig. S8). However, natural fire regimes have been altered by human impacts to the point that few natural fire regimes currently exist, which makes it challenging to project future wildfire potential under a changing climate (Abatzoglou and Williams 2016, Parisien et al. 2016, Mansuy et al. 2019).

Extreme weather events are not events to which species can quickly adapt, but rather are resilient to (Hoover et al. 2014, Isbell et al. 2015). Species could be resilient by shifting their range in response to climate change, but this can lead to positive feedback. Pecl et al. (2017) found that, in response to climate change, mountain pine beetles exhibited resiliency by shifting their range to higher elevations and northwards from the Rocky Mountains into British Columbia. They are now exploiting new host trees that have fewer defenses, which will ultimately alter the species composition and vegetation communities within wilderness areas (Cudmore et al. 2010, Pecl et al. 2017). If currently existing species and ecosystems are to continue to survive within wilderness areas by adapting to climate change, they will need to be managed to maintain or increase their resilience to environmental change, which presents challenges to wilderness managers (Appel 2015).

All our mid-century projections of extreme weather events indicate the availability (i.e., dissimilarity index values) and position (i.e., forward and backward velocities) of current climate conditions will change for species and ecosystems within wilderness areas through shifts in current conditions and formation of novel conditions (Fig. 4). Because the boundaries of wilderness areas are static, the ability of species and ecosystems to disperse, shift, or adapt with current climate conditions will be vital (Garcia et al. 2014, Elsen et al. 2020). The slower the projected forward and backward climate velocities, or the lower the dissimilarity index value, the more likely species will be able to adapt or have the time to adapt to changes in their environment (Garcia et al. 2014). Ordonez et al. (2014) and Loarie et al. (2009) both projected that the velocity of climate change will vary geographically and will be faster than historic species migration rates in some areas. Montane landscapes, which are high elevation areas with cold and wet environments are projected to have species migration rates within the range of historic migration rates and be well represented within protected areas in the future (Loarie et al. 2009, Elsen et al. 2020). Many wilderness areas in the western U.S. include montane landscapes and species occurring in these wilderness areas may have time to adapt to future changes in climate velocity and conditions (Loarie et al. 2009, Aycrigg et al. 2013, Elsen et al. 2018).

Climate projections inherently contain uncertainty (Suggitt et al. 2017, Belote et al. 2018). Additional uncertainty arises from our understanding of how species and ecosystems will respond to differing velocities of climate change, different climate conditions, and the potential feedbacks from surrounding land-use changes (Garcia et al. 2014, Langdon and Lawler 2015, Michalak et al. 2018). Even though there is uncertainty within our projections, our results indicate that all wilderness areas will undergo changes in both the availability and position of current climate conditions, but the degree of change varies geographically and may be driven by the ecological integrity of individual wilderness areas (Belote et al. 2018, Michalak et al. 2018).

Our future projections of climate change within wilderness areas will challenge wilderness users and managers to rethink the concept of “untrammeled,” which was used to describe how to manage wilderness areas within The Wilderness Act (U.S. Public Law 88-577). Management actions for adapting to climate change include restraint (i.e., passive management or acceptance of change), increasing resilience by reducing existing stressors (i.e., habitat fragmentation), resist change (i.e., active management to increase resilience), and realignment that directs or facilitates change (i.e., assisted migration; Aplet and Cole 2010, Hobbs et al. 2010, Stephenson and Millar 2012, Long and Biber 2014, Aplet and Mckinley 2017). Restraint is viewed as the default management action for wilderness areas but could include acquiring land adjacent to wilderness areas to reduce the influence of LULC change and to increase landscape connectivity (Stephenson and Millar 2012, Martin and Watson 2016, Belote et al. 2017b). This approach could reduce the loss of core habitat we projected for wilderness areas. Even though The Wilderness Act establishes and protects wilderness areas, there is flexibility within the law to permit active management within wilderness areas (Long and Biber 2014). In fact, during 2011–2015, 37% of 527 wilderness units underwent management actions that most commonly included vegetation treatments, wildfire control, and wildlife restoration projects (Lieberman et al. 2018). We projected that all wilderness areas would have an increase in the frequency of wildfire weather, therefore, management actions to reduce the frequency of wildfires could prove vital to maintaining the existing vegetation communities.
within wilderness areas. However, active management to increase resilience and resist change are only short-term strategies but could allow time to develop long-term strategies for realignment (Stephenson and Millar 2012, Lugo 2014, Appel 2015). As our analysis has demonstrated, wilderness managers will need multiple strategies with built-in flexibility to tackle how climate change will influence wilderness areas. But any strategy to address climate change adaptation in wilderness areas needs to be informed by monitoring of ecological processes and changes in climate metrics, such as magnitude and timing of change (Hobbs et al. 2010, Garcia et al. 2014, Belote et al. 2017b).

In the future, wilderness areas undergoing shifts in current climate conditions and projected to undergo more frequent extreme weather events, such as drought and wildfires, may not protect or even comprise the ecological diversity and processes for which they were originally designated to protect (Aycrigg et al. 2016, Parks et al. 2018, Mansuy et al. 2019). If our societal goal is to sustain current ecological conditions, then wilderness areas may need to be actively managed to be resilient to climate change (Stephenson and Millar 2012, Long and Biber 2014). But if our societal goal is to practice restraint and allow wilderness areas and the species, ecosystems and processes therein, to adapt to changing climatic conditions then we should anticipate a departure from current ecological conditions (Aplet and Mckinley 2017). The later goal allows for wilderness areas to act as areas of baseline change or untreated control landscapes against which all other change can be gauged (i.e., land-use change; Belote et al. 2015). This goal is considered the “observation” zone within the portfolio approach described by Aplet and Mckinley (2017), in which wilderness areas are the building blocks of future species assemblages and ecological conditions. The “observation” zone retains background rates of climate and ecological change without unintentional outcomes from maladaptive management to address resilience (Aplet and Mckinley 2017). Our climate projections imply that wilderness users and managers need to have ongoing discussions on the future management of wilderness areas and potential tradeoffs in maintaining ecological condition and untrammeled character of the landscape under changing climate conditions.

**Ranking of wilderness areas**

Ultimately, we want our results to inform wilderness area management and preservation by providing a landscape context for strategizing future conservation planning using projections of land use, land cover, and climate change in and around wilderness areas. To inform conservation planning of wilderness areas, we ranked wilderness areas by land use, land cover, and climate change into three bins using our multivariate metric of future change (Figs. 5, 6). Even though many wilderness areas, in both scenarios, fell into the middle bin (i.e., between 33 and 67) individual wilderness areas within this bin have high decile values, on which the multivariate metric of future change is based, for one or more LULC and/or climate change variable(s). This information is important for wilderness managers to know what future changes to address at each wilderness area. Our results not only provide a landscape context for conservation planning across the NWPS, but also informs future conservation planning for individual wilderness areas throughout the NWPS.

There are up to 32 wilderness areas ranked >67 under the high scenario based on our multivariate metric of future change, which warrant concern by conservation planners and wilderness managers because they are projected to undergo the most change (Figs. 5, 6, Table 1). Furthermore, concern about these wilderness areas is warranted because interactions between land use and climate have been shown to explain on average 54% of the variation in species richness, species composition, and ecosystem processes, which was greater than climate or land-use change alone (Peters et al. 2019). Even though, nationwide, there are fewer wilderness areas in the eastern U.S., eastern wilderness areas are projected to undergo more land-use and climate changes than other regions of the U.S. (Fig. 5d, Table 1). Additionally, the speed of climate and land-use change is projected to be faster in the eastern U.S. compared with other regions (Ordonez et al. 2014). Because wilderness areas in the eastern U.S. are projected to have an increase in land-use change in surrounding areas (Fig. 5d), this could lead to indirect intervention that alters species diversity and ecological processes, and which would make these wilderness areas ineffective as “observation” zones in the portfolio approach proposed by Aplet and Mckinley (2017). Therefore, even within the NWPS there may be wilderness areas along a continuum of sustaining their ecological integrity into the future. Strategies for conservation planning will need to consider changes in both land use and climate, particularly in the wilderness areas in the eastern U.S.

At the current trajectory of observed climate change (e.g., increase in temperature), it would require >70% reduction in carbon emissions by 2050 to limit the rise in temperature to 2°C (Walsh et al. 2014). Even if emissions are reduced by this ambitious target, climate change will impact wilderness areas for decades or centuries (Elsen et al. 2020, Tierney et al. 2020). In contrast, national, state, and local policies can influence the rate of land-use and land cover change immediately (Radeloff et al. 2012, Hamilton et al. 2013, Martinuzzi et al. 2013, 2015). For instance, policies directed at increasing connectivity between wilderness areas will reduce the loss of core habitat related to land use and land cover change, which in turn is related to biodiversity loss (Keely et al. 2018, Wildlife Corridors Conservation Act 2019, Hilty et al. 2020). These types of policies could also benefit areas with a projected increase in agricultural land use, which we expected would be one of the greatest land-use
changes in areas surrounding wilderness by mid-century. We also projected an increase in development and clearcutting near wilderness areas, but wilderness areas and the ecosystem services they provide could benefit from existing land-use policies that prevent these projected increases. Our results project that the area surrounding wilderness areas will be influenced by changes in land use, but existing and potentially new policies (Wildlife Corridors Conservation Act 2019) can reduce that influence to ensure the future integrity of wilderness areas.

**Conclusions**

As our projections of land use, land cover, and climate change have indicated, wilderness areas are not immune to change, but effective management of wilderness areas necessitates an understanding of the influence of these changes on the ecological characteristics and processes for which wilderness areas were established. Wilderness managers are charged with the stewardship of wilderness areas and walk the fine line between maintaining ecological conditions and untrammeled character of wilderness. Our projections provide wilderness managers with the broader context within which each individual wilderness area is being influenced by land use, land cover, and climate change to adjust management strategies, whether that be intensive management intervention, managing for anticipated future conditions, or maintaining wilderness areas as untreated controls. It is this information that is needed to ensure effective wilderness management under changing environmental conditions and the continued presence of wilderness in the future.

**Acknowledgments**

We thank Susan Fox, Beth Hahn, and Sean Parks for their valuable input throughout this project. Funding was provided by the Aldo Leopold Wilderness Research Institute through USFS Agreement No. 17JV11221639050. We thank Volker Radeloff and Dave Helmers at the SILVIS Laboratory in the Department of Forest and Wildlife Ecology at the University of Wisconsin and Andy Allstadt at the U.S. Fish and Wildlife Service for sharing their knowledge and data. Input from an anonymous reviewer and James Watson is appreciated. The authors declare no conflicts of interest. JLA, TRM, RTB, and SM conceived of the research idea and design. JLA and TRM carried out the research. TRM conducted the analyses. JLA, TRM, RTB, and SM wrote, revised, and reviewed the manuscript.

**Literature Cited**

Abatzoglou, J. T., and A. P. Williams. 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences of the United States of America 113:11770–11775.

Allen, C. D., and D. D. Breshears. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings of the National Academy of Sciences of the United States of America 95:14839–14842.

Allstadt, A. J., S. J. Vavrus, P. J. Heglund, A. M. Pidgeon, W. E. Thomgartian, and V. C. Radeloff. 2015. Spring plant phenology and false springs in the conterminous US during the 21st century. Environmental Research Letters 10:104008.

Aplet, G. H., and D. N. Cole. 2010. The trouble with naturalness: rethinking park and wilderness goals. Pages 12–29 in D. N. Cole and L. Yung, editors Beyond naturalness: rethinking park and wilderness stewardship in an era of rapid change. Island Press, Washington D.C., USA.

Aplet, G. H., and P. S. Mckinley. 2017. A portfolio approach to managing ecological risks of global change. Ecosystem Health and Sustainability 3:e01261.

Appel, P. A. 2015. Planning for adaptation and restoration in wilderness. George Washington Journal of Energy and Environmental Law 6:52–59.

Augspurger, C. K. 2013. Reconstructing patterns of temperature, phenology, and frost damage over 124 years: spring damage risk is increasing. Ecology 94:41–50.

Aycrigg, J. L., A. Davidson, L. K. Svancara, K. J. Gergely, A. McKerrow, and J. M. Scott. 2013. Representation of ecological systems within the protected areas network of the Continental United States. PLoS ONE 8:e0054689.

Aycrigg, J. L., J. Tricker, R. T. Belote, M. S. Dietz, L. Duarte, and G. H. Aplet. 2016. The next 50 years: opportunities for diversifying the ecological representation of the National Wilderness Preservation System within the contiguous United States. Journal of Forestry 114:396–404.

Baldwin, R., and K. Beazley. 2019. Emerging paradigms for biodiversity and protected areas. Land 8:43.

Barbero, R., J. T. Abatzoglou, N. K. Larkin, C. A. Kolden, and B. Stocks. 2015. Climate change presents increased potential for very large fires in the contiguous United States. International Journal of Wildland Fire 24:892–899.

Belote, R. T., et al. 2017a. Wild, connected, and diverse: building a more resilient system of protected areas. Ecological Applications 27:1050–1056.

Belote, R. T., et al. 2017b. Mapping conservation strategies under a changing climate. BioScience 67:494–497.

Belote, R. T., C. Carroll, S. Martimuzzi, J. Michalak, J. W. Williams, M. A. Williamson, and G. H. Aplet. 2018. Assessing agreement among alternative climate change projections to inform conservation recommendations in the contiguous United States. Scientific Reports 8:1–13.

Belote, R. T., M. S. Dietz, and G. H. Aplet. 2015. Allocating untreated “controls” in the National Wilderness Preservation System as a climate adaptation strategy: a case study from the Flathead National Forest, Montana. Northwest Science 89:239–254.

Belote, R. T., M. S. Dietz, B. H. McRae, D. M. Theobald, M. L. McClure, G. H. Irwin, P. S. McKinley, J. A. Gage, and G. H. Aplet. 2016. Identifying corridors among large protected areas in the United States. PLoS ONE 11:e0154223.

Belote, R. T., and M. B. Wilson. 2020. Delineating greater ecosystems around protected areas to guide conservation. Conservation Science and Practice 2:1–10.

Blois, J. L., P. L. Zarnetske, M. C. Fitzpatrick, and S. Finnegan. 2013. Climate change and the past, present, and future of biotic interactions. Science 341:499–504.

Brooks, T. M., et al. 2002. Habitat loss and extinction in the hotspots of biodiversity. Conservation Biology 16:909–923.

Buote, P. C., J. A. Hicke, H. K. Preisler, J. T. Abatzoglou, K. F. Raffa, and J. A. Logan. 2017. Recent and future climate suitability for whitebark pine mortality from mountain pine beetles varies across the western US. Forest Ecology and Management 399:132–142.
Carroll, C., J. J. Lawler, D. R. Roberts, and A. Hamann. 2015. Biotic and climatic velocity identify contrasting areas of vulnerability to climate change. PLoS ONE 10:e0140486.

Convention on Biological Diversity. 2020. Update of the zero draft of the post-2020 global biodiversity framework. https://www.cbd.int/article/zero-draft-update-august-2020

Cudmore, T. J., N. Bjørklund, A. L. Carroll, and B. Staffan Lindgren. 2010. Climate change and range expansion of an aggressive bark beetle: evidence of higher beetle reproduction in naive host tree populations. Journal of Applied Ecology 47:1036–1043.

Dawson, C. P., and J. C. Hendee. 2009. Wilderness management: stewardship and protection of resources and values. Fourth edition. Fulcrum Publishing, Golden, Colorado, USA.

Di Marco, M., S. Ferrier, T. D. Harwood, A. J. Hoskins, and J. E. M. Watson. 2019. Wilderness areas halve the extinction risk of terrestrial biodiversity. Nature 573:582–585.

Elsen, P. R., W. B. Monahan, E. R. Dougherty, and A. M. Merenlender. 2020. Keeping pace with climate change in global terrestrial protected areas. Science Advances 6:eaa80184.

Elsen, P. R., W. B. Monahan, and A. M. Merenlender. 2018. Global patterns of protection of elevational gradients in mountain ranges. Proceedings of the National Academy of Sciences of the United States of America 115:6004–6009.

Foley, J. A., et al. 2005. Global consequences of land use. Science 309:570–574.

Garcia, R. A., M. Cabeza, C. Rahbek, and M. B. Araújo. 2014. Multiple dimensions of climate change and their implications for biodiversity. Science 344:1247579.

Gaston, K. J., S. F. Jackson, L. Cantú-Salazar, and G. Cruz-Piñón. 2008. The ecological performance of protected areas. Annual Review of Ecology, Evolution, and Systematics 39:93–113.

Gimmi, U., S. L. Schmidt, T. J. Hawbaker, C. Alcántara, U. Gafvert, and V. C. Radeloff. 2011. Increasing development in the surroundings of U.S. National Park Service holdings jeopardizes park effectiveness. Journal of Environmental Management 92:229–239.

Goetz, S. J., P. Jantz, and C. A. Iantz. 2009. Connectivity of core habitat in the Northeastern United States: parks and protected areas in a landscape context. Remote Sensing of Environment 113:1421–1429.

Haddad, N. M., et al. 2015. Habitat fragmentation and its lasting impact on Earth’s ecosystems Science Advances 1:e1500902.

Hamilton, C. M., S. Martinuzzi, A. J. Plantinga, V. C. Radeloff, D. J. Lewis, W. E. Thogmartin, P. J. Heglund, and A. M. Piedgeon. 2013. Current and future land use around a Nationwide Protected Area Network. PLoS ONE 8:e6055737.

Hansen, A. J., and R. Defries. 2007. Ecological mechanisms linking protected areas to surrounding lands. Ecological Applications 17:974–988.

Hayhoe, K., J. Edmonds, R. E. Kopp, A. N. LeGrande, B. M. Sanderson, M. F. Wehner, and D. J. Wuebbles. 2017. Climate models, scenarios, and projections. Pages 133–160 in D. J. Wuebbles, D. W. Fahey, K. A. Hibbard, D. J. Dokken, B. C. Stewart, and T. K. Maycock, editors. Climate Change Special Report: Fourth National Climate Assessment, Volume I. U.S. Global Change Research Program, Washington, DC, USA. https://doi.org/10.7930/J0WEH2N54

Hilty, J., et al. 2020. Guidelines for conserving connectivity through ecological networks and corridors. https://doi.org/10.2305/fu nc.h.2020.pag.30.en

Hobbs, R. J., et al. 2010. Guiding concepts for park and wilderness stewardship in an era of global environmental change. Frontiers in Ecology and the Environment 8:483–490.

Hoffmann, S., and C. Beierkuhnlein. 2020. Climate change exposure and vulnerability of the global protected area estate from an international perspective. Diversity and Distributions 26:1496–1509.

Hoffmann, S., S. D. H. Irl, and C. Beierkuhnlein. 2019. Predicted climate shifts within terrestrial protected areas worldwide. Nature Communications 10. https://doi.org/10.1038/s41467-019-12603-w

Holsinger, L., S. A. Parks, M. Parisien, C. Miller, E. Batllori, and M. A. Moritz. 2019. Climate change likely to reshape vegetation in North America’s largest protected areas. Conservation Science and Practice 1. https://doi.org/10.1111/csp2.50

Hoover, D. L., A. K. Knapp, and M. D. Smith. 2014. Resistance and resilience of a grassland ecosystem to climate extremes. Ecology 95:2646–2656.

Huang, C., S. N. Goward, J. G. Masek, N. Thomas, Z. Zhu, and J. E. Vogelmann. 2010. An automated approach for reconstructing recent forest disturbance history using dense Landsat time series stacks. Remote Sensing and Environment 114:183–198.

Inouye, D. W. 2000. The ecological and evolutionary significance of frost in the context of climate change. Ecology Letters 3:457–463.

Inouye, D. W. 2008. Effects of climate change on phenology, frost damage, and floral abundance of montane wildflowers. Ecology 89:353–362.

IPCC. 2000. Land use, land-use change, and forestry. A Special Report of the Intergovernmental Panel on Climate Change. https://www.ipcc.ch/site/assets/uploads/2018/03/srl-en-1.pdf

IPCC. 2012. Managing the risks of extreme events and disasters to advance climate change adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change. https://www.ipcc.ch/report/managing-the-risks-of-extreme-events-and-disasters-to-advance-climate-change-adaptation/

Iribarne, J. J., and M. G. Byram. 1968. A drought index for forest fire control. Research Paper SE-38. U.S. Department of Agriculture Forest Service, Southeast Forest Experimental Station, Ashville, North Carolina, USA. http://www.treesearch.fs.fed.us/pubs/40

Kral-O’Brien, K. C., P. L. O’Brien, and J. P. Harmon. 2019. Need for false spring research in the Northern Great Plains, USA. Agricultural and Forest Meteorology 4:190025.

Lambin, E. F., H. J. Geist, and E. Lepers. 2003. Dynamics of land-use and land-cover change in tropical regions. Annual Review of Environment and Resources 28:205–241.

Langdon, J. G. R., and J. J. Lawler. 2015. Assessing the impacts of projected climate change on biodiversity in the protected areas of western North America. Ecosphere 6. https://doi.org/10.1890/14-04010.1

Lieberman, L. B., H. Hahn, and P. Landres. 2018. Manipulating the wild: a survey of restoration and management interventions in U.S. wilderness. Restoration Ecology 26:900–908.

Loarie, S. R., P. B. Duffy, H. Hamilton, G. P. Asner, C. B. Field, and D. D. Ackerly. 2009. The velocity of climate change. Nature 462:1052–1055. 
U.S. Geological Survey (USGS) Gap Analysis Project (GAP). 2016. Protected Areas Database of the United States (PAD-US), version 1.4. www.usgs.gov
van Vuuren, D. P., and T. R. Carter. 2014. Climate and socio-economic scenarios for climate change research and assessment: reconciling the new with the old. Climatic Change 122:415–429.
Venter, O., et al. 2016. Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. Nature Communications 7. https://doi.org/10.1038/ncomms12558
Vogt, P., and K. Riitters. 2017. GuidosToolbox: universal digital image object analysis. European Journal of Remote Sensing 50:352–361.
Walsh, J., et al. 2014. Chapter 2 Our changing climate. Pages 19–67 in J. M. Melillo, T. C. Richmond, and G. W. Yohe, editors. Climate change impacts in the United States: The Third National Climate Assessment. U.S. Global Change Research Program. https://doi.org/10.7930/J0KW5CXT
Watson, J. E. M., D. F. Shanahan, M. Di Marco, J. Allan, W. F. Laurance, E. W. Sanderson, B. Mackey, and O. Venter. 2016. Catastrophic declines in wilderness areas undermine global environment targets. Current Biology 26:2929–2934.
Watson, J. E. M., O. Venter, J. Lee, K. R. Jones, J. G. Robinson, H. P. Possingham, and J. R. Allan. 2018. Protect the last of the wild. Nature 563:27–30.
Wildlife Corridor Conservation Act. 2019. S. 1488 – Wildlife Corridors Conservation Act of 2019. https://www.congress.gov/bill/116th-congress/senate-bill/1499
Williams, C. M., H. A. L. Henry, and B. J. Sinclair. 2015. Cold truths: how winter drives responses of terrestrial organisms to climate change. Biological Reviews 90:214–235.
Wilson, T. S., B. M. Sleeter, and A. W. Davis. 2015. Potential future land use threats to California’s protected areas. Regional Environmental Change 15:1051–1064.
Zhang, F., Q. Quan, F. Ma, D. Tian, D. L. Hoover, Q. Zhou, and S. Niu. 2019. When does extreme drought elicit extreme ecological responses? Journal of Ecology 107:2553–2563.

Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.2471/full