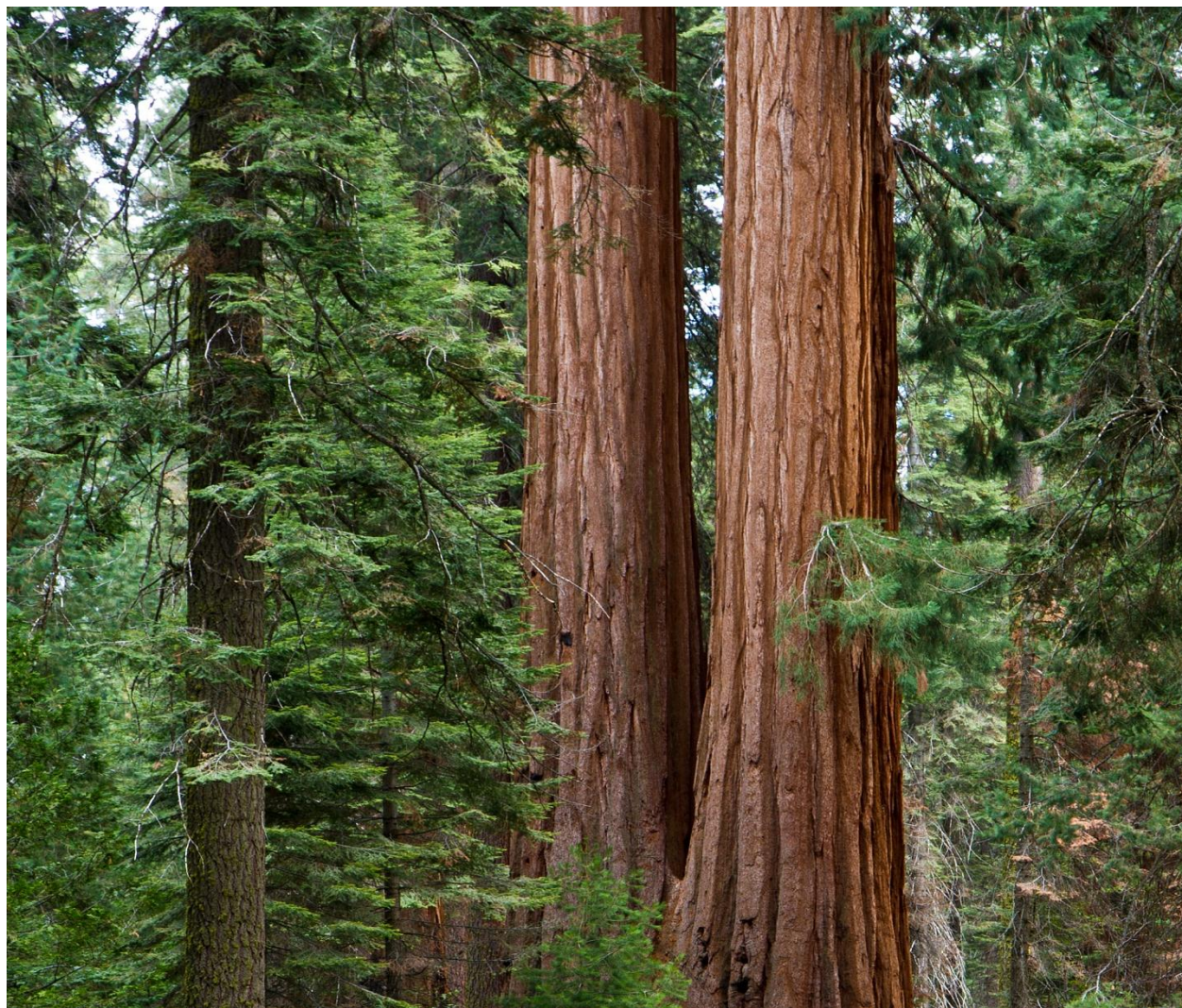




A Natural Resource Condition Assessment for Sequoia and Kings Canyon National Parks

Appendix 12 – Intact Forests

Natural Resource Report NPS/SEKI/ NRR—2013/665.12



ON THE COVER

Giant Forest, Sequoia National Park
Photography by: Brent Paull

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Appendix 12 – Intact Forests

Natural Resource Report NPS/SEKI/ NRR—2013/665.12

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Fort Collins, Colorado

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Scope of the Analysis

Sequoia-Kings Canyon National Parks (SEKI) encompass striking physical gradients that include more than 4,000 m of relief and the rain shadow effect of the Great Western Divide (Stephenson 1998, Urban et al. 2000). As a result, the landscape ranges from barren peaks to massive conifer forests to dry shrublands. Despite this diversity, forests are the most common vegetation type. In SEKI, they account for $\frac{1}{3}$ of the area and almost $\frac{1}{2}$ of the non-barren lands (USGS-NPS Vegetation Mapping Program 2007). For this Natural Resource Condition Assessment (NRCA), we have divided the SEKI forests into two focal resource categories: Foothill Vegetation and Intact Forests. Foothill vegetation includes the oak-dominated forests that occur largely below 1,830 m elevation (Rodriguez-Buritica and Suding, NRCA assessment). Intact forests consist of the expansive conifer-dominated ecosystems that occur in the montane habitats of SEKI. These forests, along with their angiosperm affiliates, cover 97,746 ha and account for 84% of the area classified as forest (USGS-NPS 2007, Map 1). For this analysis, a forest is defined as a vegetated landscape with trees over 5 m tall, with their crowns interlocking, and generally forming 60-100% cover (NatureServe 2004). Intact refers to forests that have not been harvested or otherwise developed since European settlement.

Critical Questions

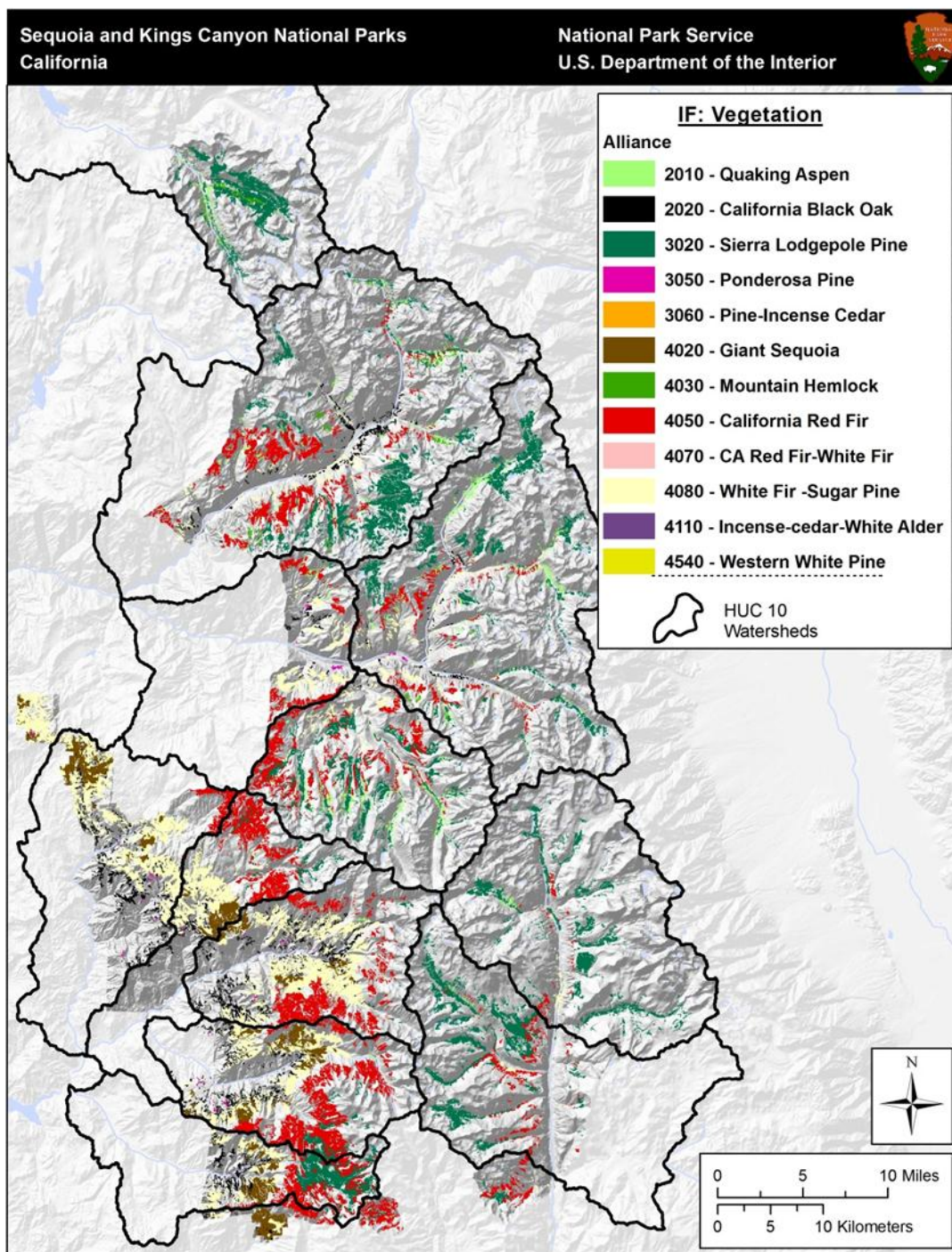
The critical question for intact forests at SEKI is how “intact” are they? The concept of ecological integrity has proven to be a useful framework to assess the condition of protected ecosystems (Parrish et al. 2003, Tierney et al. 2009, Reza and Abdullah 2011). In broad terms, ecological integrity can be defined as the capability of the ecosystem to maintain the composition and structure of the biological community; the productive capacity of the ecosystem; and the ability to recover from stress and disturbance. Given this definition, it is clear that integrity shares an intellectual history with ideas expressed in terms of ecosystem health, ecological sustainability, and resilience (Rapport et al. 1998). The distinguishing feature of integrity is its explicit focus on informing management decisions and evaluating conservation efforts.

Recently Tierney et al. (2009) evaluated the structure, composition, and function of forests in the National Parks of the northeastern United States in terms of ecological integrity. Their work provides a template for the condition assessment of the SEKI forests. Thus our critical questions focus on addressing the key components of integrity as identified in Tierney et al. (2009):

Does the landscape structure in terms of patch size and continuity support species that rely on habitats found in the interior of forests?

Do the intact forests recruit a sufficient abundance of large-diameter trees (live and dead) that are essential to maintaining structural complexity and habitat heterogeneity?

Are the intact forests self-sustaining and productive?



Map 1. Distribution of intact forest alliances in Sequoia Kings Canyon National Parks. Data source: SEKI vegetation map.

The challenge is that these components of integrity must be measured in a meaningful and practical manner (Reza and Abdullah 2011). Forest ecosystems are complex multifaceted phenomena that integrate biological and physical gradients across space and time. Although it is necessary to reduce this complexity in order to inform management, the results of this simplification are indices that, at their best, capture relative changes. Thus it is important to develop benchmarks for comparison. Traditionally this baseline has been defined as the range of natural variation or historic conditions (Landres et al. 1999). In the absence of such information, the metrics should distinguish an impacted or degraded state from a relatively unimpaired and functioning state (Tierney et al. 2009).

In addition to the assessment of ecological integrity, we also estimated the magnitude of known stressors on the intact forest. In planning for the NRCA, we identified the altered fire regime, air pollution (ozone and nitrogen deposition), exotic pests, and drought as the primary threats. By combining a measure of condition (ecological integrity) along with an evaluation of vulnerability (magnitude of known stressors), we hoped to provide insights into the current and future status of the intact forests at SEKI.

Data Sources and Types Used in Analysis

We used six sources of information for this assessment.

1. The SEKI vegetation map (USGS-NPS 2007).

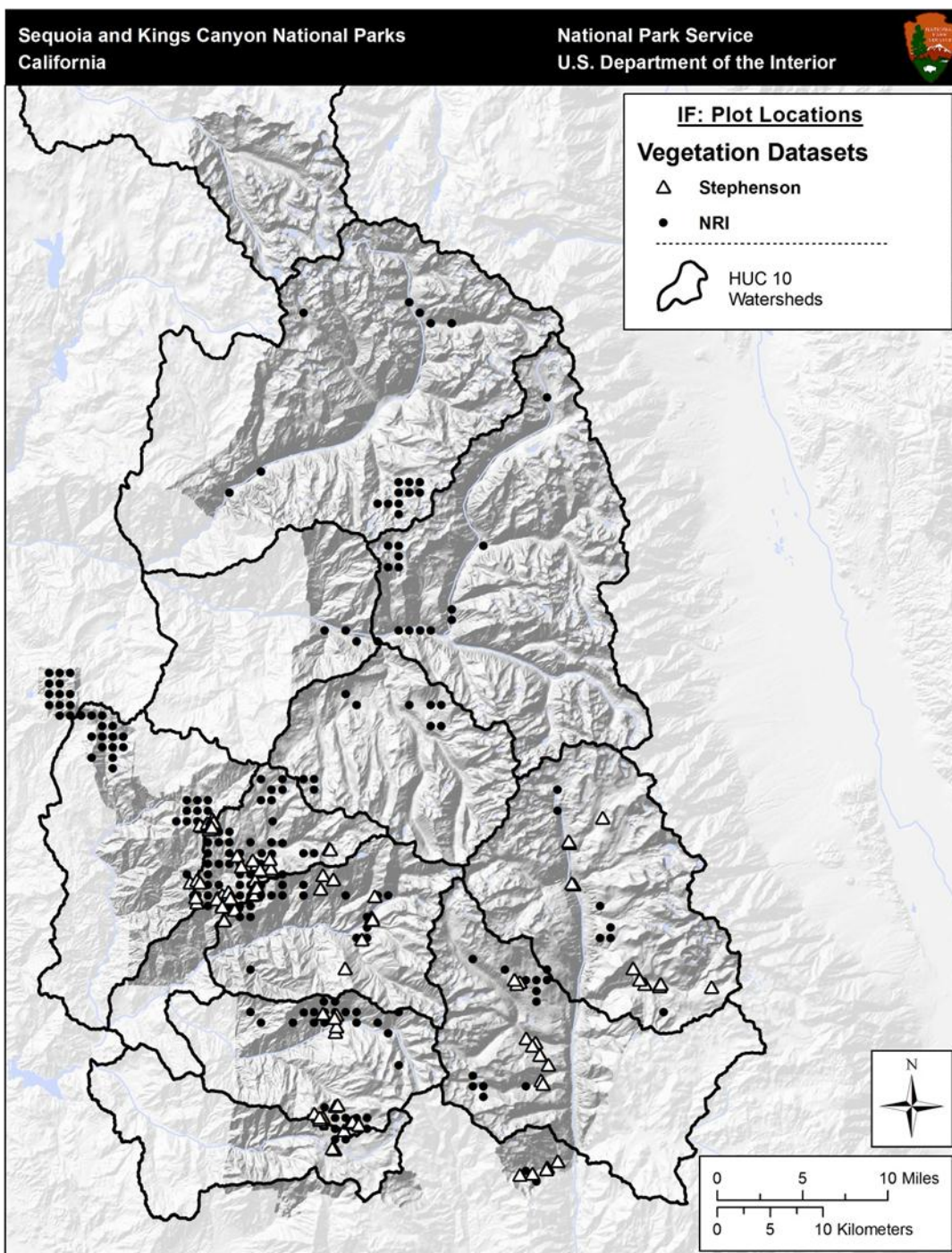
The vegetation map for SEKI provides a spatially explicit and hierarchical classification scheme that incorporates the physiognomy of the landscape and the species composition of the plant community. It follows the national standard for vegetation classification and was adapted from mapping schemes initially developed for Yosemite National Park (NatureServe 2004). Wall-to-wall spatial coverage is based on the photointerpretation of high resolution imagery that was refined and validated with an extensive field sampling effort. The final map for SEKI was completed in 2007 (USGS-NPS 2007). Analysis of landscape-level metrics on patch size and continuity was based on the maps results. Moreover the map provides the means to scale plot-level results and to interpolate what is effectively point data to the entire Park. In short, the map was the key spatial integrator of this resource assessment.

2. Data from the natural resource inventory (NRI) for SEKI (Graber and Haultain 1993).

In 1985, the National Park Service initiated a plot-based parkwide survey of the vascular plants. Plot locations were selected according to a stratified random sampling scheme using the 1-km UTM coordinate grid (Map 2). Each 1 km UTM intersection represented a potential sample point; clusters of these points were visited throughout the two parks in such a way as to maximize efficiency and to provide broad geographical coverage of slope, elevation, and aspect classes present in the parks. Data were collected from 0.1 ha circular plots (17.8 m radius). Within each plot, the diameter at breast height (dbh) was recorded for all trees taller than 1.4 m and with a dbh of 1 cm or more. In extremely dense plots, diameters were estimated. A total of 628 plots were established between 1985 and 1996. A small subset was measured more than once.

Using the location of the plot center, we attributed each plot with the characteristics of its co-located SEKI vegetation map polygon. Thus each NRI plot was assigned to a physiognomic and vegetation class. The NRI data represents the most detailed, unbiased, and comprehensive information on the status of the vegetation in SEKI. It formed the informational backbone for this assessment of intact forests.

3. The published results from the long-term forest demography plots (Riegal et al. 1988, van Mantgem and Stephenson 2007).



Map 2. Location of the NRI and Stephenson plots in Sequoia-Kings Canyon National Parks that were classified as being in the intact forest. Data source: NRI and Stephenson vegetation gradient.

Beginning in 1982, permanent plots were established to describe the structure and composition of old-growth Sierran mixed conifer forests and to monitor long-term changes in this ecosystem (Riegel et al. 1998). Every tree ≥ 1.37 m tall was identified to species and dbh was measured. All live trees were mapped along with additional qualitative information (e.g., canopy class, vigor,

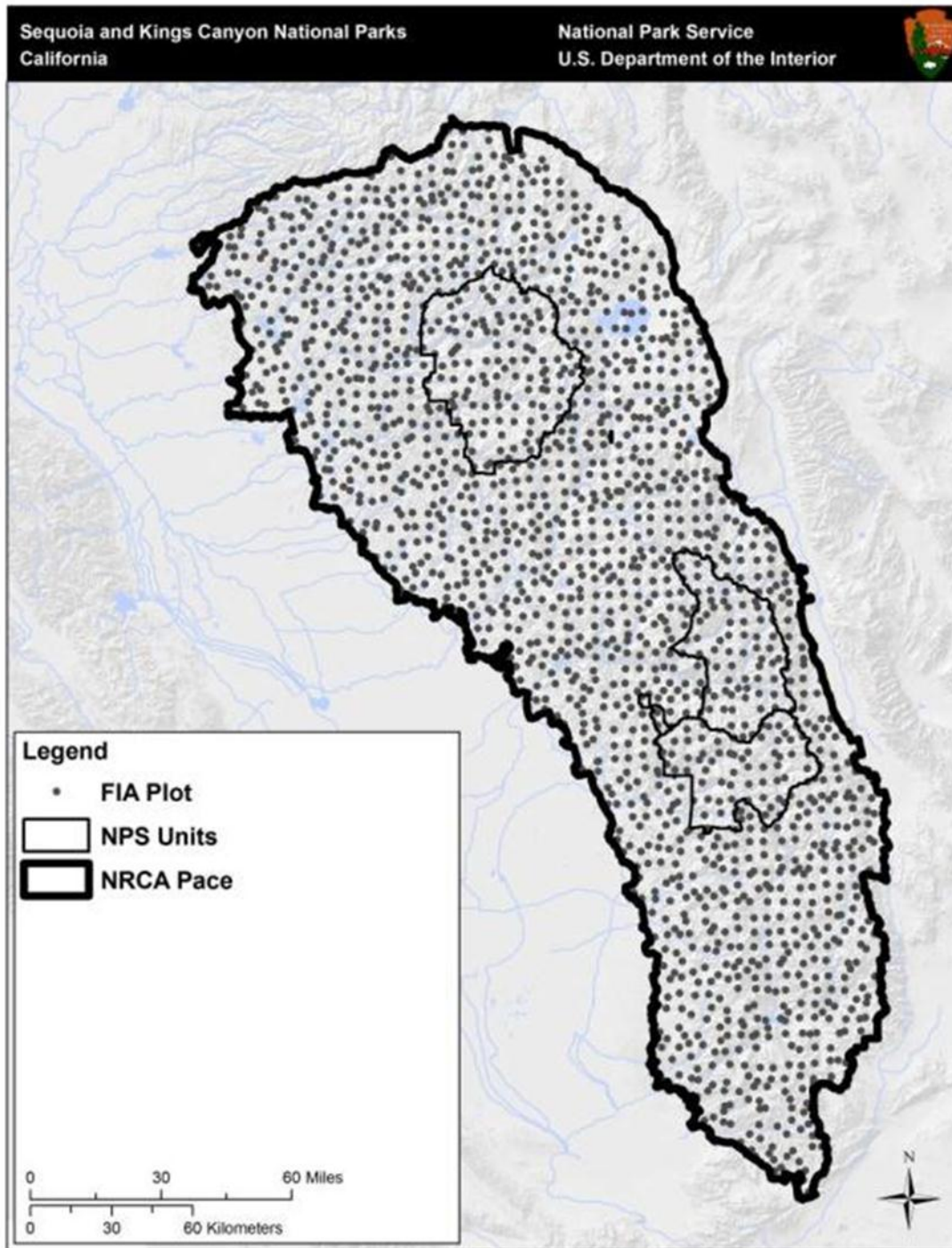
and disturbance). After an initial effort to establish six plots in SEKI (1982-1983), the network has been expanded to mixed conifer stands throughout SEKI and Yosemite National Parks (YOSE). As of 2007, there were 21 plots that varied in size from 1 to 2.5 ha. As part of the monitoring effort, plots are checked annually for tree mortality and recruitment. Trees that died were evaluated for factors associated with death, including an inventory of physical damage and signs of tree-killing or tree-weakening pathogens and insects. Approximately every five years, the diameters of live trees are remeasured (van Mantgem and Stephenson 2007). The demography plot network provides crucial estimates of key drivers of forest change -- namely tree growth, survival, and recruitment. For this condition assessment, we relied exclusively on published results to inform the assessment of intact forests.

4. Data from the Stephenson forest gradient plots in Sequoia National Park (Stephenson 1998).

Between 1982 and 1984, Stephenson (1998) established a network of 0.1 ha plots in the conifer forests of SEKI to measure forest change along the complex elevational gradient. Plots were located to sample a broad range of elevation, slope steepness, and aspect within two major watersheds (Kaweah River and Kern River, Map 2). All live trees ≥ 1.37 m tall were identified and assigned to dbh classes that were ultimately organized into 5-cm intervals from 0 through 140 cm and then > 140 cm (N. Stephenson, USGS WERC, personal communication). Based on its location, we attributed each plot with the characteristics of its co-located SEKI vegetation map polygon. Thus each Stephenson plot was assigned to a physiognomic and vegetation class. These data are the most comparable in terms of scope and coverage to the NRI plots. Thus our analysis of temporal trends was largely based on differences in key metrics estimated using the two data sets.

5. Data from the USDA Forest Service's Forest Inventory and Analysis (FIA) program (<http://fia.fs.fed.us/>).

The USDA Forest Service conducts a national inventory of forest resources (i.e. FIA) that includes a network of field plots distributed at a density of approximately one plot per 2,492 ha. Measurements in these plots (Phase 2) include nested sampling of trees (species identification, dbh, status, canopy position). In the western United States, FIA plots are measured on a ten-year cycle. The fraction measured in any individual year is used as a representative sample of all plots in the region. Details on the design of the inventory and access to the data are available online (<http://fia.fs.fed.us/>). From the 2010 FIA database, we downloaded the data for all plots in the boundaries of the protected area centered ecosystem (PACE) that encompasses SEKI (Map 3).



Map 3. Distribution of Forest Inventory and Analysis plots in the protected area centered ecosystem that encompasses Sequoia Kings Canyon National Parks. Date source: USDA FIA program.

We then screened the data for the most recent measurement for all annual plots in the PACE. There were a total of 951 plots measured since FIA revised their protocols to measure individual trees (2001). These data represent a contemporary and regional perspective on the status of the

southern Sierran montane forest. Our analysis of this data provided context for interpreting the status and integrity of the SEKI intact forests.

6. Data from the survey of white pine blister rust in SEKI (Duriscoe and Duriscoe 2002).

Between 1995 and 1999, the incidence of white pine blister rust (*Cronartium ribicola*) was measured across the ranges of the susceptible tree species (the five-needled pines: sugar pine, western white pine, foxtail pine, whitebark pine and limber pine) in SEKI. The complete inventory included 154 plots with 50 susceptible trees in each plot. See Duriscoe and Duriscoe (2002) for details on the sampling strategy. Based on the location of the blister rust plots, we assigned a vegetation alliance to each plot. For this analysis, we included all plots that contained sugar pine or western white pine and were located in an intact forest alliance. Sugar and western white pine accounted for approximately 8% of aboveground live tree biomass in the intact forest. The other three pines only occur in abundance in the higher elevation woodlands (SEKI vegetation map). They constituted a rare component of the intact forest (Fig. 1). Thus we excluded the rare species in the intact forest assessment. The status of the five-needled pines in SEKI is addressed in a separate chapter. These data provided the means to quantify blister rust infection in a spatially explicit fashion.

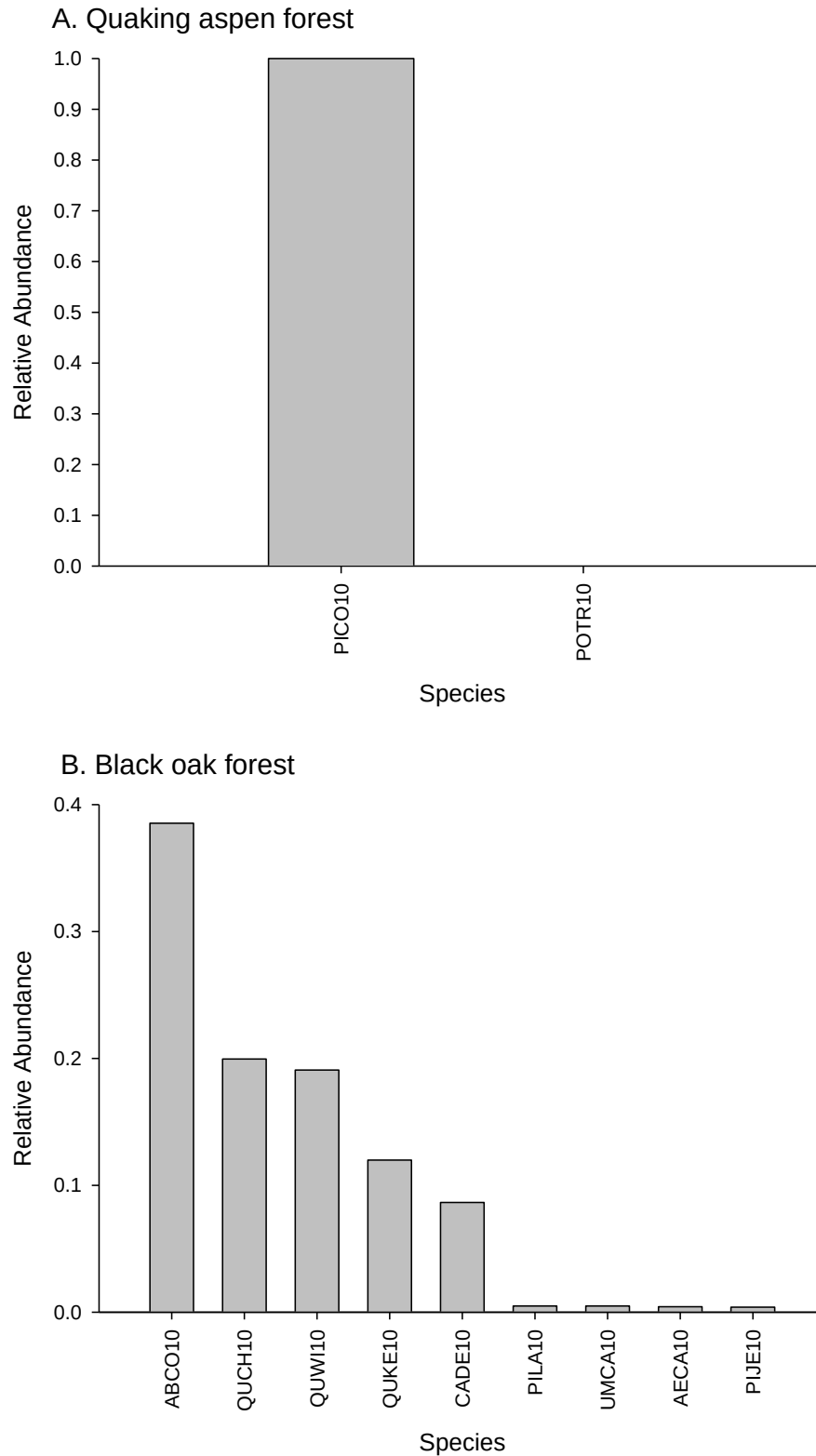


Figure 1.A-B. Dominance-diversity curves for the intact forest alliances of A) quaking aspen forest and B) black oak forest. Results based on the Natural Resource Inventory. Species codes listed in Appendix 2. Relative abundance is based on aboveground biomass for trees with a diameter > 12.7 cm.

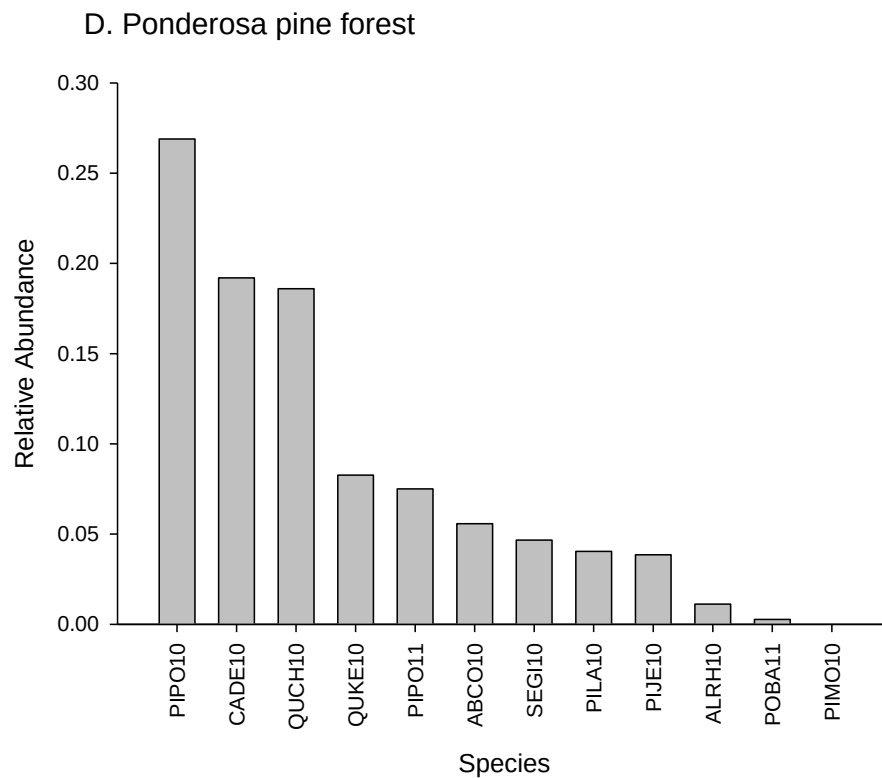
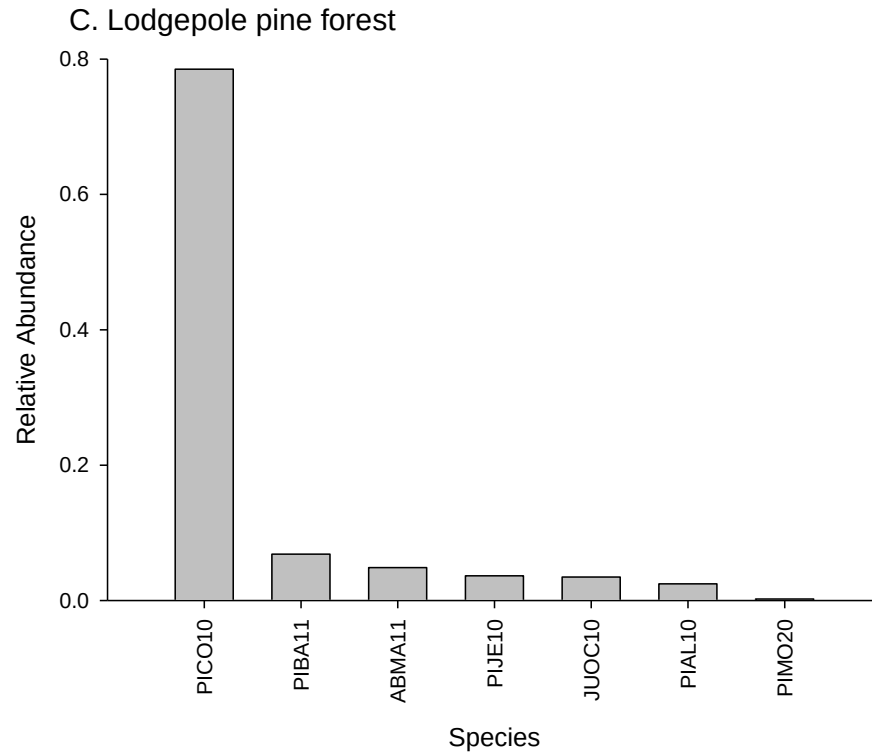


Figure 1.C-D. Dominance-diversity curves for the intact forest alliances of E) lodgepole pine forest and D) Ponderosa pine forest. Results based on the Natural Resource Inventory. Species codes listed in Appendix 2. Relative abundance is based on aboveground biomass for trees with a diameter > 12.7 cm.

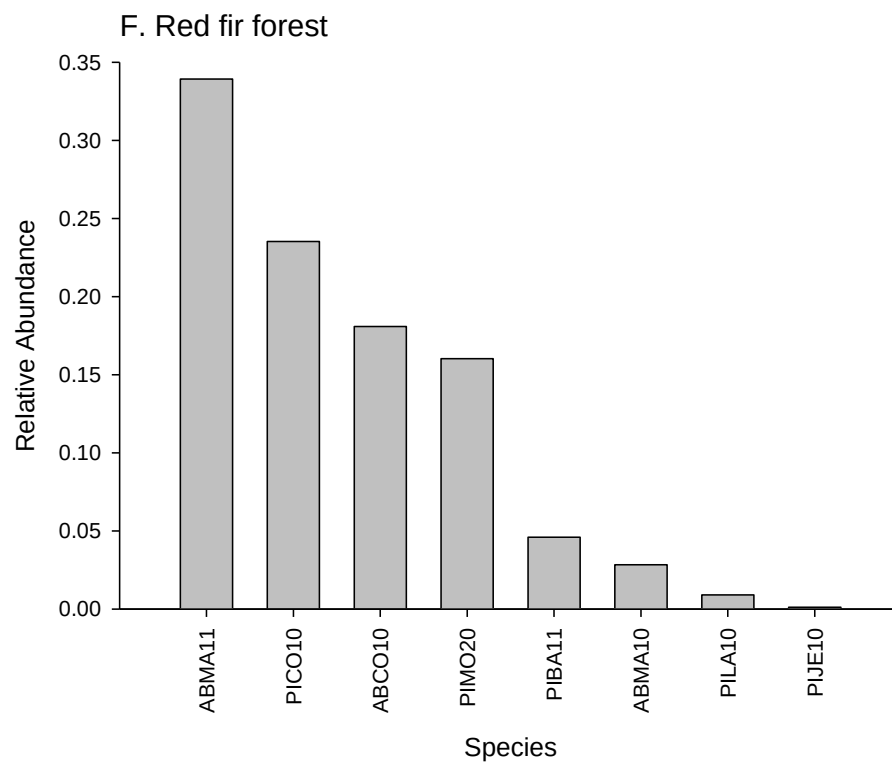
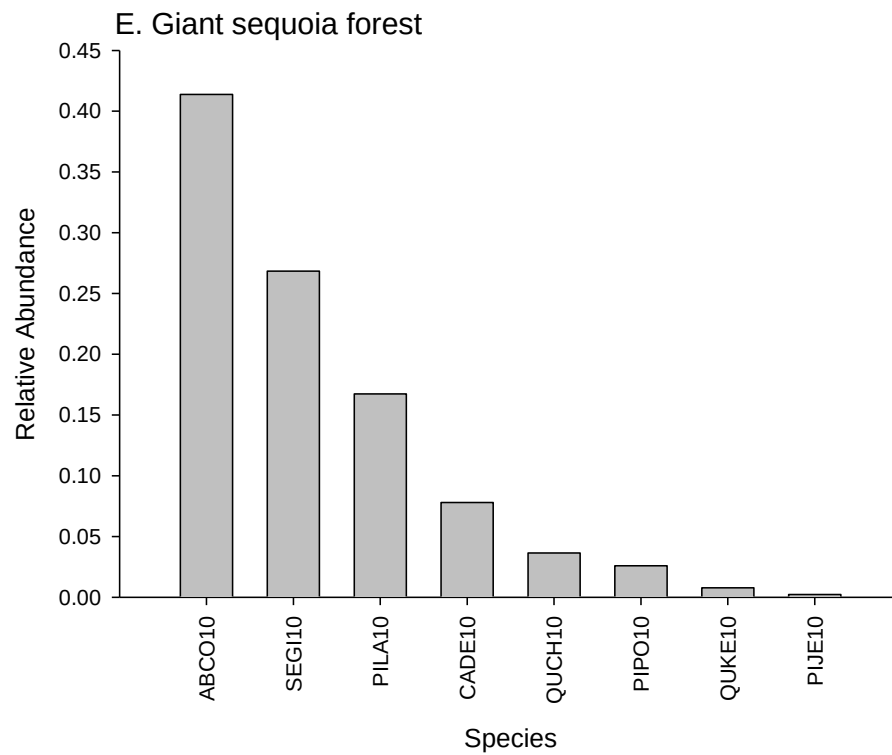


Figure 1.E-F. Dominance-diversity curves for the intact forest alliances of E) giant sequoia forest and F) red fir forest. Results based on the Natural Resource Inventory. Species codes listed in Appendix 2. Relative abundance is based on aboveground biomass for trees with a diameter > 12.7 cm.

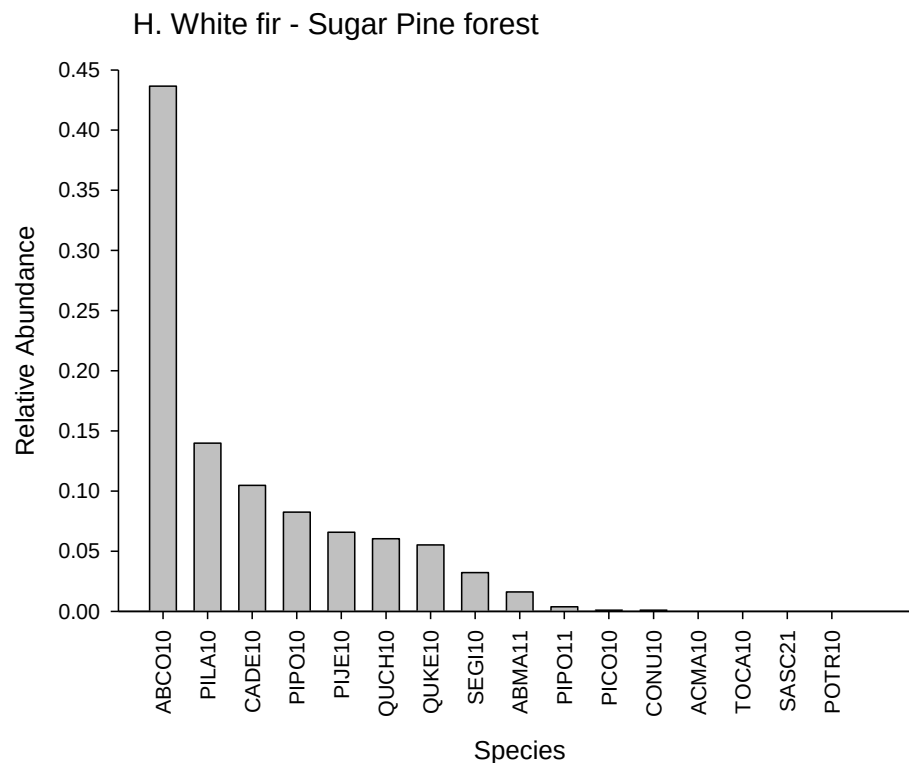
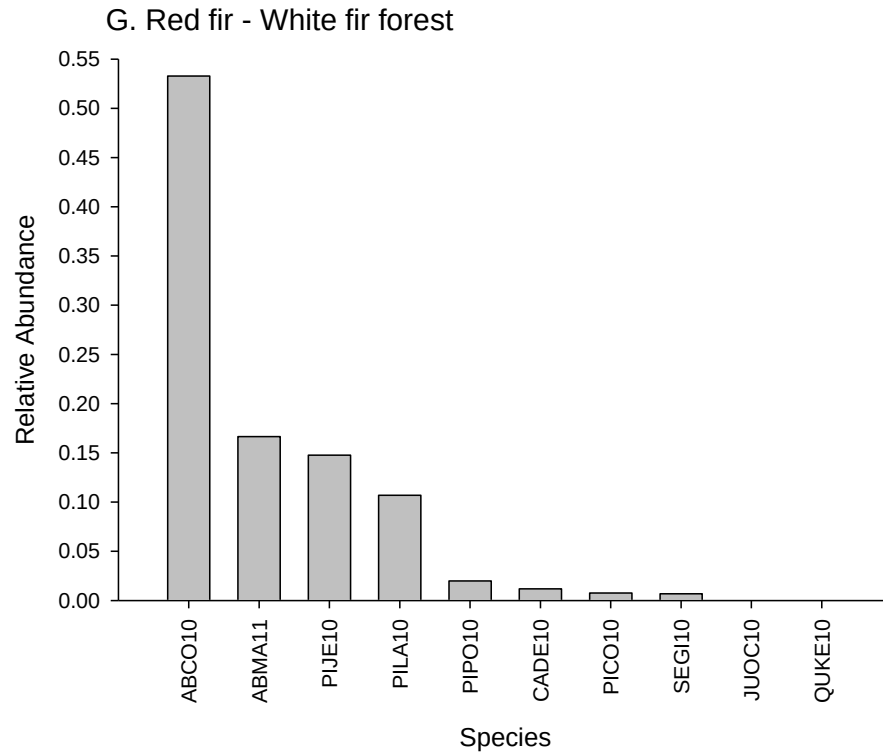


Figure 1.G-H. Dominance-diversity curves for the intact forest alliances of G) red fir – white fir forest and H) white fir – sugar pine forest. Results based on the Natural Resource Inventory. Species codes listed in Appendix 2. Relative abundance is based on aboveground biomass for trees with a diameter > 12.7 cm.

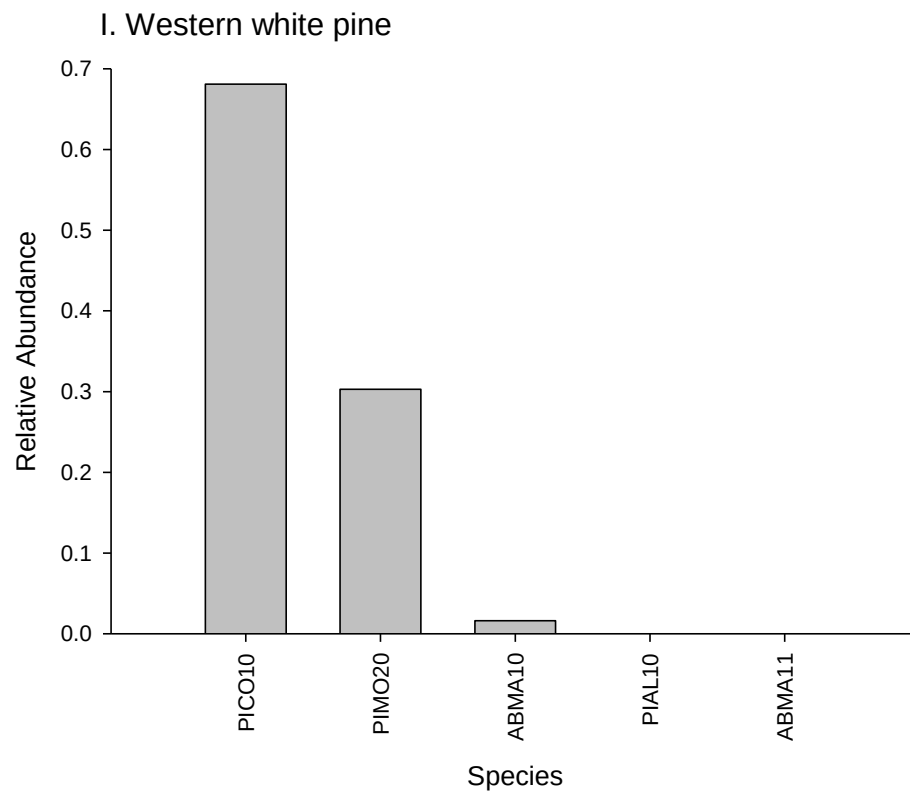


Figure 1.I. Dominance-diversity curves for the intact forest alliances for western white pine. Results based on the Natural Resource Inventory. Species codes listed in Appendix 2. Relative abundance is based on aboveground biomass for trees with a diameter > 12.7 cm.

Reference Conditions

Intact Forest Composition and Structure

Introduction

The intact forests of SEKI represent the diversity of the extensive montane vegetation that occurs along the slopes of the Sierra Nevada (Fites-Kaufman et al. 2007). Although our ultimate goal is an integrated, spatially explicit assessment of this focal resource, the first step was to describe and quantify the constituent elements. To meet this objective, we summarized key forest metrics of the major forest alliances (as defined in the SEKI vegetation map) using the NRI data.

There were 12 forest alliances designated as intact forests (Table 1, Appendix 1, Map 1). These alliances correspond to forest types defined for the Sierra Nevada (Keeler-Wolf 2007, Fites-Kaufman et al. 2007). In the broadest sense, they can be described as the Sierran mixed conifer and true fir forests with two hardwood dominated patches (California black oak and quaking aspen) interspersed among the conifers. Note: throughout this report, we used the well-accepted and widely used common names for the tree species in the text. To ease integration among diverse data sets, we retained the 6-character species code established by SEKI in the tables and figures. Appendix 2 provides a comprehensive taxonomic reference for all tree species included in this report.

Methods

We chose commonly reported metrics to describe the intact forests (forest extent, tree density, tree basal area, species composition, species richness) as well as measurements that inform the evaluation of ecological integrity. We divided components of integrity into three categories: landscape structure, forest structure and composition, and function (adapted from Tierney et al. 2009). We then developed measures to assess each category.

At the alliance level, landscape structure was estimated by the mean size and frequency of patches on the landscape for each alliance. Patches were defined as the mapping units (minimum size = 0.5 ha) used in the SEKI vegetation map. Our analysis included the 3,349 polygons classified as forest (median size = 8.5 ha). We described the composition of each alliance by creating dominance-diversity curves that rank species by abundance. We used the size (as measured by dbh) and identity of all trees recorded in the NRI plots to develop the curves. We also calculated the diameter distribution of trees by size class in order to gain insight into the dynamics of the forest (Zenner 2005, Coomes and Allen 2007).

Big trees are widely recognized as important forest constituents. They are indicative of productive older forests with intact disturbance regimes. They contribute both structural complexity and key wildlife habitat (Franklin and Fites-Kaufman 1996, van Pelt and Sillett 2008, Lutz et al. 2009). We used the species-specific definitions of large-diameter trees developed by Lutz et al. (2009) from the historical data available for comparable vegetation in Yosemite National Park (YOSE, Table 2). We calculated the density of big trees for each NRI plot.

Table 1. Summary of extent and composition of the intact forest alliances in Sequoia-Kings Canyon National Parks. The vegetation extent follows the Final Mapping Classification (USGS-NPS 2007). Composition based on National Resource Inventory data (Graber and Haultain 1993). Totals reported for “Area”, “Frequency” and “Tree Species Richness”. Means with standard errors in parentheses reported for all other metrics. The sample size (n) represents the number of plots in Alliance; 4-digit number represents Alliance code. Only trees ≥ 12.7 cm dbh were included in the calculations.

Alliance	Total Area (ha)	Frequency (# polygons)	Mean Size (ha)	Density (stems ha ⁻¹)	Big Tree Density (stems ha ⁻¹)	Basal Area (m ² ha ⁻¹)	Aboveground Biomass (Mg ha ⁻¹)	Tree Species Richness
Quaking aspen (n=3, 2010)	2,492	993	2.5 (0.1)	112 (111)	0	1.5 (1.8)	3.3 (4.1)	2
Black oak (n=5, 2020)	5,040	1,325	3.8 (0.004)	590 (71)	40 (12)	41.4 (15.2)	250.7 (86.0)	9
Lodgepole pine (n=48, 3020)	20,478	4,340	4.7 (0.1)	467 (70)	31 (5)	32.3 (3.7)	177.5 (27.0)	7
Ponderosa pine (n=1, woodland, 3050)	198	50	4.0	170	10	24.6	116.0	2
Ponderosa pine (n=13, forest, 3060)	5,539	538	10.3 (0.5)	777 (164)	42 (8)	46.5 (9.0)	254.0 (52.5)	12
Giant sequoia (n=21, 4020)	4,613	126	36.6 (8.8)	700 (176)	46 (6)	101.1 (18.4)	895.5 (220.9)	8
Mountain hemlock (n=0, 4030)	561	133	4.2 (0.4)	--	--	--	--	--
Red fir (n=20, 4050)	19,973	1,956	10.2 (0.5)	461 (67)	30 (6)	50.2 (5.3)	274.4 (32.9)	8
Red fir-White fir (n=22, 4070)	12,967	890	14.6 (1.0)	561 (121)	35 (5)	44.2 (6.1)	281.4 (47.6)	10
White fir-Sugar pine (n=63, 4080)	16,062	1,294	12.4 (1.5)	631 (64)	34 (3)	51.6 (4.0)	338.2 (30.8)	16
Incense cedar (n=0, 4111)	132	22	6.0 (1.3)	--	--	--	--	--
Western white pine-Lodgepole (n=8, 4540)	10,252	934	11.0 (0.5)	321 (51)	15 (6)	21.0 (4.6)	93.5 (22.7)	5

Table 2. Comparisons of big tree density. The definition of “big tree” is based on Lutz et al. (2009). Estimates for Yosemite National Park (YOSE) were taken from Lutz et al. 2009. SEKI-NRI estimates from Natural Resources Inventory (NRI) data for Sequoia-Kings Canyon National Parks. SEKI-FIA estimates were from Forest Inventory and Analysis (FIA) plots in SEKI. Estimates for the Protected-Area Centered Ecosystem (PACE) were based on recent FIA plots in the PACE.

Genus	species	DBH limit (cm)	YOSE 1988-99 (stems ha ⁻¹)	SEKI- NRI 1985-96 (stems ha ⁻¹)	SEKI-FIA 2001- 2009 (stems ha ⁻¹)	PACE- FIA 2001-09 (stems ha ⁻¹)	YOSE v SEKI- NRI (%)	SEKI NRI v FIA (%)	SEKI- FIA v PACE (%)
Abies	concolor	≥92	6.5	9	7	2	28	-22	-71
Abies	magnifica	≥92	17.8	23	18	5	23	-22	-72
Calocedrus	decurrens	≥92	3.9	3	2	1	-30	-33	-50
Juniperus	occidentalis	≥92	4.1	0	1	1	inf	inf	0
Pinus	albicaulis	≥31	33.6	14	22	10	-140	57	-55
Pinus	contorta	≥61	16.6	14	12	8	-19	-14	-33
Pinus	jeffreyi	≥92	5.3	13	2	2	59	-85	0
Pinus	lambertiana	≥92	11.1	14	0	0	21	-100	inf
Pinus	monticola	≥92	7.9	6	1	2	-32	-83	100
Pinus	ponderosa	≥92	6.9	6	0	0	-15	-100	inf
Pseudotsuga	menziesii	≥92	3.6	np	np	0	nc	nc	nc
Quercus	chrysolepis	≥31	35.8	17	26	10	-111	53	-62
Quercus	kelloggii	≥31	26.7	13	6	12	-105	-54	100
Tsuga	mertensiana	≥61	21.8	90	np	np	76	nc	nc

np= species not present in sample; nc = no comparison; inf = infinite comparison.

Forest biomass accumulation provides a valuable synoptic perspective on ecosystem function (Bormann and Likens 1979). It represents a measure of both the productivity of the primary producers as well the pool of “organic capital” available to fuel heterotrophic processes (DeAngelis et al. 1981). We focused on estimating the aboveground live tree biomass. While it is not the only pool of biomass in the ecosystem, live trees constitute a large, dynamic pool that can be measured relatively accurately. Moreover it is directly linked to forest tree demography and carbon cycling (Fahey et al. 2005, van Doorn et al. In press). To estimate tree biomass, we followed protocols developed by the USDA Forest Service Forest Inventory and Analysis program (FIA 2009a). Aboveground tree biomass included estimates of the bole, bark, and crown and was calculated from FIA regional volume and biomass functions (FIA 2009b, FIA 2009c). These functions provide species-specific estimates of tree mass from dbh measurements and height estimates. Only trees ≥ 12.7 cm (5 inches) dbh were included in biomass calculations. For mature conifer forests in the Sierra Nevada, trees larger than 12.7 cm dbh typically account for more than 97% of the live tree biomass (Gonzalez et al. 2010).

Tree species richness was calculated as the total number of species found in each alliance. Only trees ≥ 12.7 cm dbh were included. Note that sampling effort (i.e., number of plots in each alliance) varied by alliance. Since species richness is sensitive to sampling effort (Palmer 1990), comparisons across alliances with low sample size (e.g., <10 plots) must be made with caution.

Results/Discussion

The intact forests of SEKI encompassed a wide range of communities (Table 1). The most abundant alliance was lodgepole pine with more than 20,000 ha in SEKI closely followed by the red fir alliance (19,973 ha). All of the alliances, even ones distinguished by their hardwood component, were dominated by conifer species (Fig. 1). Tree species richness varied from a low of two in quaking aspen to 16 in white fir-sugar pine. Most alliances showed the typical dominance-diversity curves expected for temperate forests, namely that a handful of common tree species monopolize the resources with relatively few rare species (Fig. 1). The lodgepole pine alliance was an exception. It had strong monodominance – lodgepole pine accounted for approximately 80% of the aboveground live tree biomass (Fig. 1C).

The abundance of trees declined exponentially from the smallest to largest diameter class and was consistent for all alliances (Fig. 2A). This pattern has repeatedly been reported for montane forest specifically in SEKI (e.g., Riegel et al. 1988, Parsons et al. 1992) and generally in the Sierra Nevada (e.g., Ansley and Battles 1998, Barbour et al. 2002). Typically, a negative exponential distribution of tree sizes (also referred to as a “reverse J” form) is considered a characteristic of mature (i.e., late-successional) forests (Oliver and Larson 1996). Despite limitations, recent research has confirmed the value of the using tree diameter distributions to infer stand age (Zeener 2005) and vegetational stability (Coomes and Allen 2007).

Tree density, big tree density, basal area, and biomass varied greatly among alliances (Table 1). Given this heterogeneity, measures of ecological integrity must account for the differences among alliances. We used the data from FIA plots in the PACE to develop the range of expected values. We selected plots from the comparable forest types and calculated the same values. Species composition and patterns in dominance from comparable forest types in the PACE (Fig. 3) matched those observed in the NRI plots. In terms of structure, again the results were

comparable with a tendency for the SEKI intact forests to be more massive. The SEKI forests had more big trees and greater live tree biomass (Table 3).

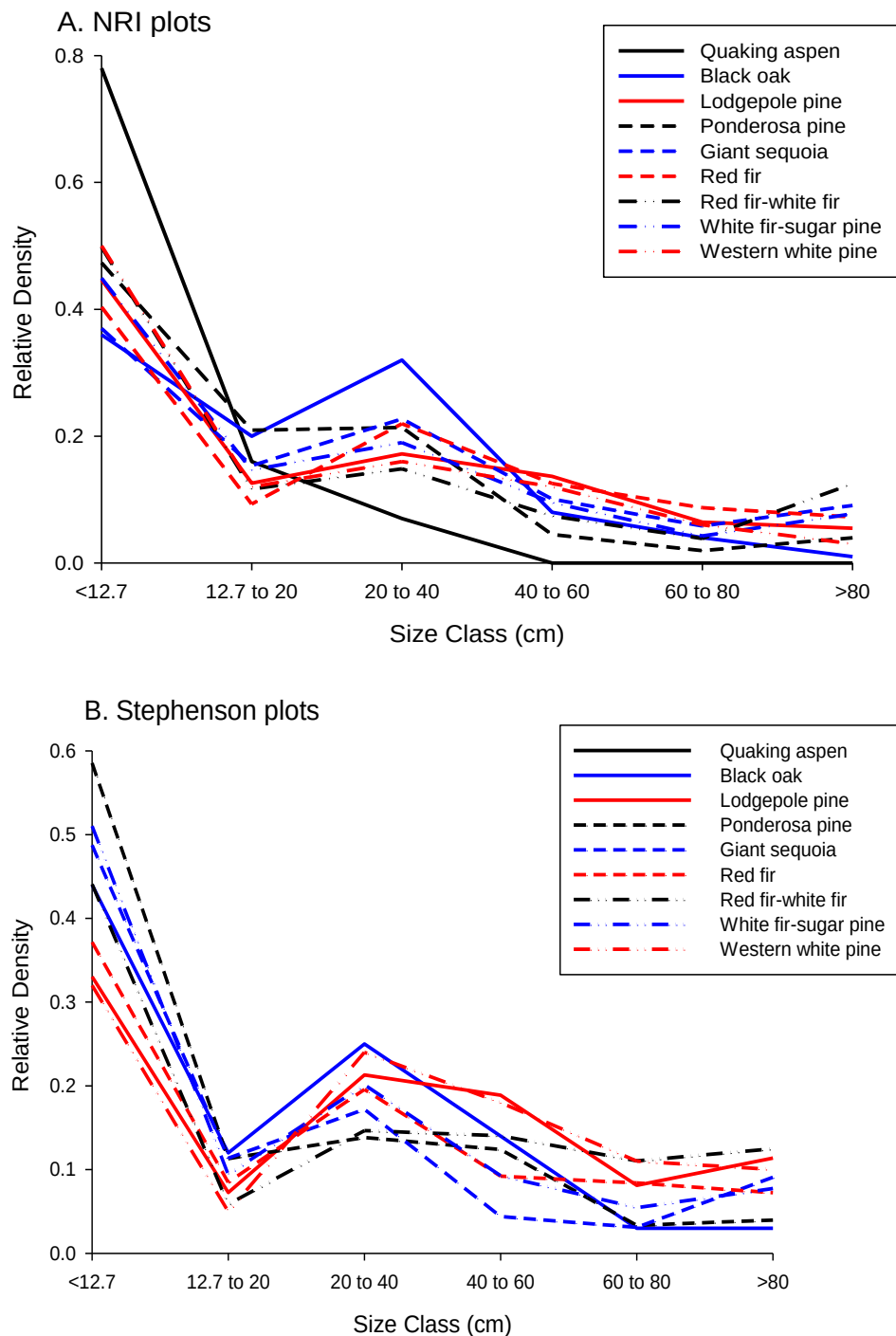


Figure 2. Size distribution by density for the intact forest alliances in Sequoia-Kings Canyon National Parks. Panel A: results from Natural Resource Inventory. Panel B: results from Stephenson Gradient Analysis. Inventories approximately 5 years apart. See text for details.

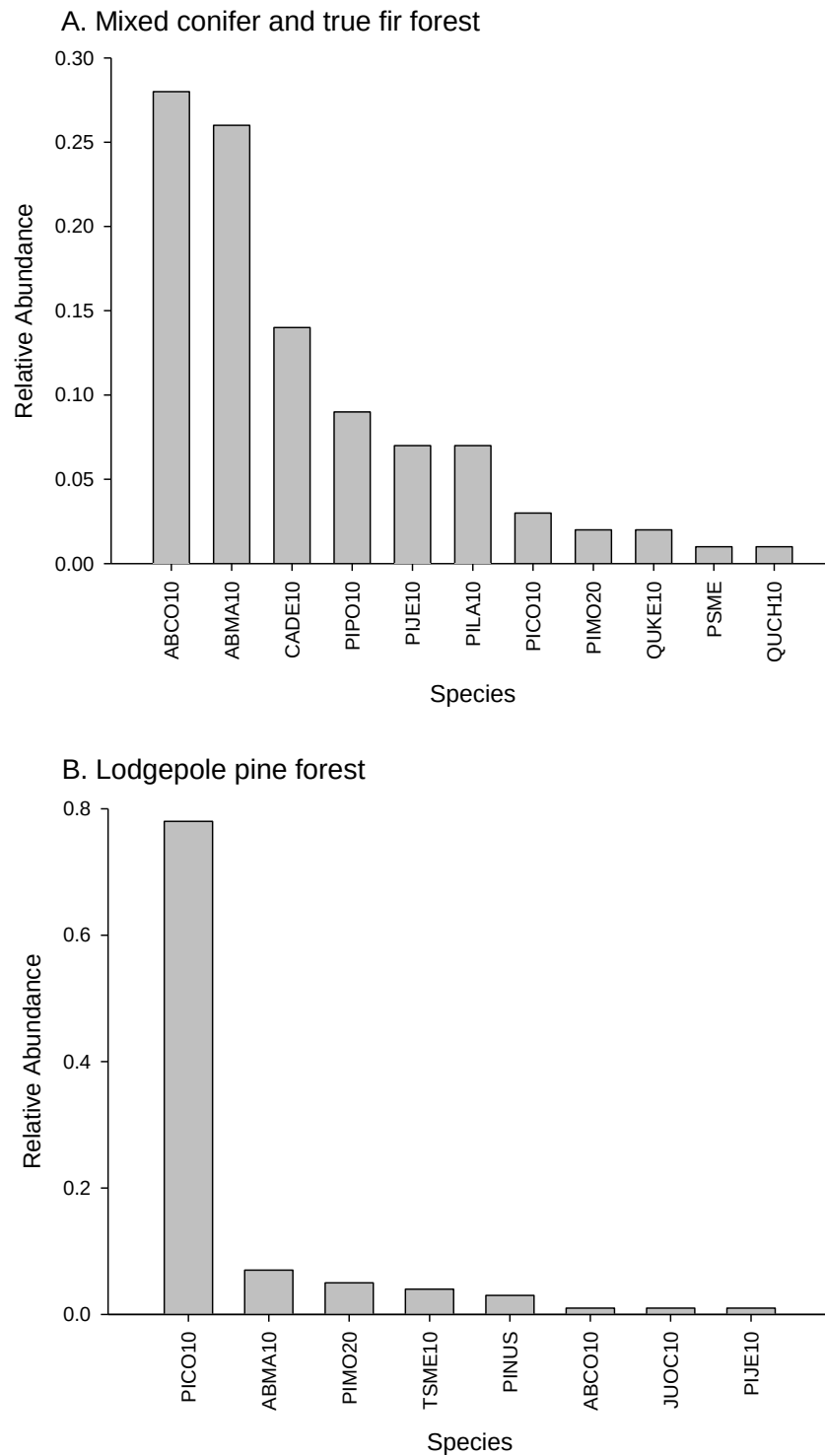


Figure 3. Dominance-diversity curves forest types in the PACE. Results based on the FIA data. Species codes listed in Appendix 2. Relative abundance is based on aboveground biomass for trees with a diameter ≥ 12.7 cm.

Table 3. Comparison of structure for two common forest types in Sequoia-Kings Canyon National Parks (SEKI). Results based on three inventories. SEKI-NRI refers to the Natural Resource Inventory conducted for SEKI (Graber and Haultain1993). SEKI-FIA refers to plots in the Forest Inventory and Analysis that were located in the Parks. PACE-FIA refers to FIA plots in the Protected-Area Centered Ecosystem that includes SEKI. Means with standard deviations in parentheses reported for all metrics. The sample size (n) represents the number of plots in the inventory. Big trees follow definition of Lutz et al. (2009). Mixed conifer/fir includes the following alliances: 4020, 4050, 4070, and 4080. Lodgepole includes only alliance 3020. PACE forest type based on a cross-walk between FIA forest types and the SEKI Final Mapping Classification. Note that the PACE-FIA plots do not include the plots in SEKI.

	SEKI-NRI (1986-1996)		SEKI-FIA (2001-2009)		PACE-FIA (2001-2009)	
	Lodgepole (n=48)	Mixed conifer/fir (n=139)	Lodgepole (n=9)	Mixed conifer/fir (n=18)	Lodgepole (n=124)	Mixed conifer/fir (n=336)
Density (stems ha ⁻¹)	467 (70)	619 (46)	166 (72)	532 (91)	615 (64)	636 (31)
Big Tree Density (stems ha ⁻¹)	31 (5)	36 (2)	23 (11)	19 (4)	24 (2)	14 (1)
Basal Area (m ² ha ⁻¹)	32.3 (3.7)	57.2 (3.9)	25.2 (14.7)	45.5 (5.8)	34.7 (1.7)	43.8 (1.2)
Aboveground Biomass (Mg ha ⁻¹)	177.5 (27.0)	396.4 (40.7)	95.2 (75.3)	256.1 (40.5)	132.4 (8.4)	215.7 (7.4)

Spatial Analyses

Our assessment of the intact forest was integrated to the watershed level. For this NRCA, the finest spatial resolution was defined by the USGS hydrological unit code 10 (HUC 10, mean size = 29,200 ha; range from 9,400 to 62,700 ha). We used the SEKI vegetation map to calculate the area-weighted contribution of each forest alliance to overall watershed average. NRI plot density was insufficient to calculate alliance means for each HUC 10 watershed (Map 4). Instead, we used the average alliance values for plots located in the next larger watershed (HUC 8). If there were no plots for an alliance in the HUC 8 watershed, we used the park-wide alliance means reported in Table 1. In effect, our spatial interpolations of the metrics used to assess the intact forest (e.g., biomass, big tree density) included differences among watersheds due to the distribution of forest alliances and the location of the watershed in one of the three major river drainages in SEKI (the Kings River, the Kern River, and the Kaweah River, Map 4).

Evaluating Ecological Integrity

Introduction

Following the model proposed by Tierney et al. (2009), we developed criteria to measure three aspects of ecological integrity: landscape structure, forest structure and composition, and ecosystem function. To obtain a spatially-explicit evaluation, we applied the criteria to the HUC 10 watersheds.

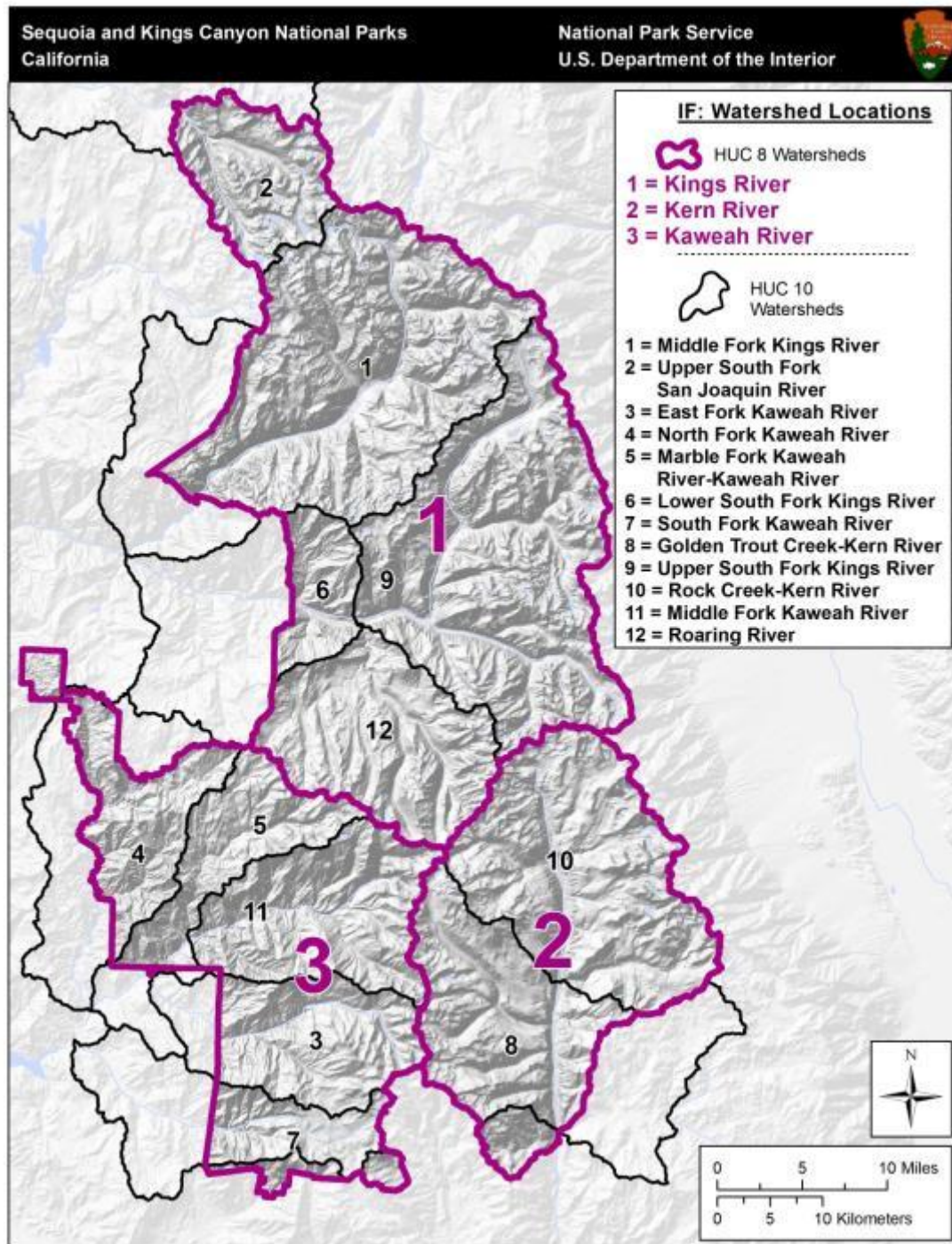
Methods

Landscape structure captures the patch size and continuity of the intact forest. In this assessment framework, larger more contiguous forest patches were considered as indicators of integrity. We calculated the area-weighted mean patch size to quantify the availability of contiguous habitat areas that are essential to the population dynamics of forest-dwelling organisms. We used the large patch index, which is simply the fraction of the total area contained in the largest patch, as a measure of continuity (McGarigal and McComb 1995). These landscape metrics were derived from the SEKI vegetation map.

Given the diversity of the intact forest in terms of species composition (Table 1), our assessment emphasized forest structure. Specifically, we focused on the density of big trees. As noted above, big trees are an essential attribute of these forests (Lutz et al. 2009). The other measure of structure was the relative density of snags (i.e., standing dead trees). Scarcity of snags is often a feature of recently disturbed or heavily managed forests. Snags represent nutrient capital that replenish coarse woody debris as they decay and ultimately fall-over. The availability of big snags are particularly important to cavity nesting birds since the large trees, with their complex canopy architecture, (van Pelt and Sillett 2008) are most likely to develop these habitat features as they decay.

We assessed two key aspects of forest function: ecosystem productivity and vegetation stability. As described above, live tree biomass has proven to be a robust index of forest function. For a particular forest type, the stand with more aboveground live tree biomass is considered to be more productive and/or least disturbed. The consistent form of the diameter distributions for all

intact forest alliances (Fig. 2A) suggests that deviations away from a negative exponential form could be used as an indicator of change.



Map 4. Location and relationship of HUC 10 and HUC 8 watersheds in Sequoia-Kings Canyon National Parks. Data source: SEKI GIS databases.

To detect changes in the diameter distribution, we used the departure index. This index is similar to the Gini coefficient used by Stohlgren (1991) to evaluate changes in size distribution of giant sequoia trees. The advantage of the departure index is that it distinguishes both the magnitude and direction of change (Menning et al. 2007) from a defined reference condition. In this case, the alliance-specific diameter-distributions presented in Fig. 2A served as the reference distributions.

For each metric, we developed a rating – good, caution, concern. We used the patch size classes suggested by Tierney et al. (2009) where an average patch size > 50 ha was considered good and < 10 ha was cause for concern (Table 4). They noted that 50 ha patches were too small to support the home range of large mammals but provided contiguous habitat for many species of birds and small mammals. Given our focus on intact forest condition in each watershed and not forest habitat in general, an emphasis on animals with more specific and smaller home ranges seemed appropriate. Largest patch index is a relative measure of fragmentation that depends on the grain of the classification (McGarigal and McComb 1995). Thus we used the observed index among our 12 watersheds to rank those more fragmented than average (Table 5). Values within one standard deviation of the mean were considered good; values between one and two standard deviations from the mean were assigned a caution rating; values less than two standard deviations from the mean were cause for concern (Table 4).

We used results from the PACE to inform expectations regarding big tree density and aboveground live tree biomass. By relying on the PACE, SEKI forests were evaluated against a regional standard. A similar strategy was used by Rocky Mountain National Park in their NRCA

NPS 2010a). We selected all the recently inventoried (2001-2010) FIA plots in PACE that were sampled from forest types similar to those defined for the intact forest. There were 512 FIA plots included in this analysis. In terms of species composition and abundance, the FIA forest types were a reasonable match to the SEKI alliances (Fig. 3, Table 3). However the PACE is several orders of magnitude larger in terms of area and plot density than the SEKI watersheds. Therefore we downscaled the PACE results using a resampling procedure. We used the mean number of NRI plots in the SEKI HUC 10 watersheds, 19, to develop downscaled-distributions of big tree density and aboveground live tree biomass. Specifically we generated 1,000 realizations of each metric by randomly sampling without replacement 19 values from the 512 FIA PACE plots. We ranked SEKI watershed values within one standard deviation of the mean as good; values more than one standard deviation but less than 2 standard deviations as caution; and values more than two standard deviations from the mean as concern (Table 4).

We adopted the criteria suggested by Tierney et al. (2009) for snag abundance (Table 4) with the exception that we set the lowest rating at a relative standard (<5% of live stems, Table 4) rather than an absolute standard. We selected the relative standard to account for the wide range of stem densities present across the intact forest alliances (Table 1). Snags were defined as standing dead trees with a diameter ≥ 12.7 cm dbh.

Table 4. Metrics and ratings for evaluating the ecological integrity of the intact forests at Sequoia-Kings Canyon National Park. See text for details.

Metric type	Metric	Rating		
		Good	Caution	Concern
Landscape structure	Weighted average patch size	>50 ha	10-50 ha	<10 ha
	Largest patch index	>10	5-10	<5
Forest structure	Big tree density	> 13 big trees ha ⁻¹	8-13 big trees ha ⁻¹	< 8 big trees ha ⁻¹
	Snag abundance (Tierney)	>10% of the standing trees are snags and > 10% of the big trees are snags	5-10% of the standing trees are snags and 5-10% of the big trees are snags	<5% of the standing trees are snags and <5% of the big trees are snags
Ecosystem function	Size distribution (Departure Index)	< 10% change	10-25% change, either direction	> 25%, either direction
	Aboveground live tree biomass	> 153 Mg ha ⁻¹	124-153 Mg ha ⁻¹	< 124 Mg ha ⁻¹

Table 5. Area-weighted means for measures of ecological integrity in the HUC 10 watersheds at SEKI. Cell color codes reflect the

HUC 10	Area-weighted patch size (ha)	Largest Patch Index (%)	Big tree density (trees ha ⁻¹)	Snags (%)	Big Snags (%)	Departure Index (-0.5, 1.5)	Aboveground live tree biomass (Mg ha ⁻¹)
Middle Fork Kings River	419	12	28	6	5	-0.01	231.4
Upper South Fork San Joaquin River	381	33	17	8	8	-0.08	130.7
East Fork Kaweah River	226	8	34	11	12	-0.01	338.8
North Fork Kaweah River	481	12	38	15	11	0.01	376.3
Marble Fork Kaweah River	544	15	33	11	13	-0.01	331.8
Lower South Fork Kings River	115	7	35	12	9	-0.02	258.8
South Fork Kaweah River	475	15	33	10	10	-0.05	335.7
Golden Trout Creek-Kern River	200	11	33	4	1	0.07	126.4
Upper South Fork Kings River	159	11	24	8	6	-0.03	187.2
Rock Creek-Kern River	97	9	34	5	2	0.09	149.1
Middle Fork Kaweah River	284	11	35	14	12	0.00	330.4
Roaring River	168	8	28	6	5	0.00	230.4

integrity rating based on criteria listed in Table 4. Coding: green = “good”; yellow= “caution”; red = “concern.”

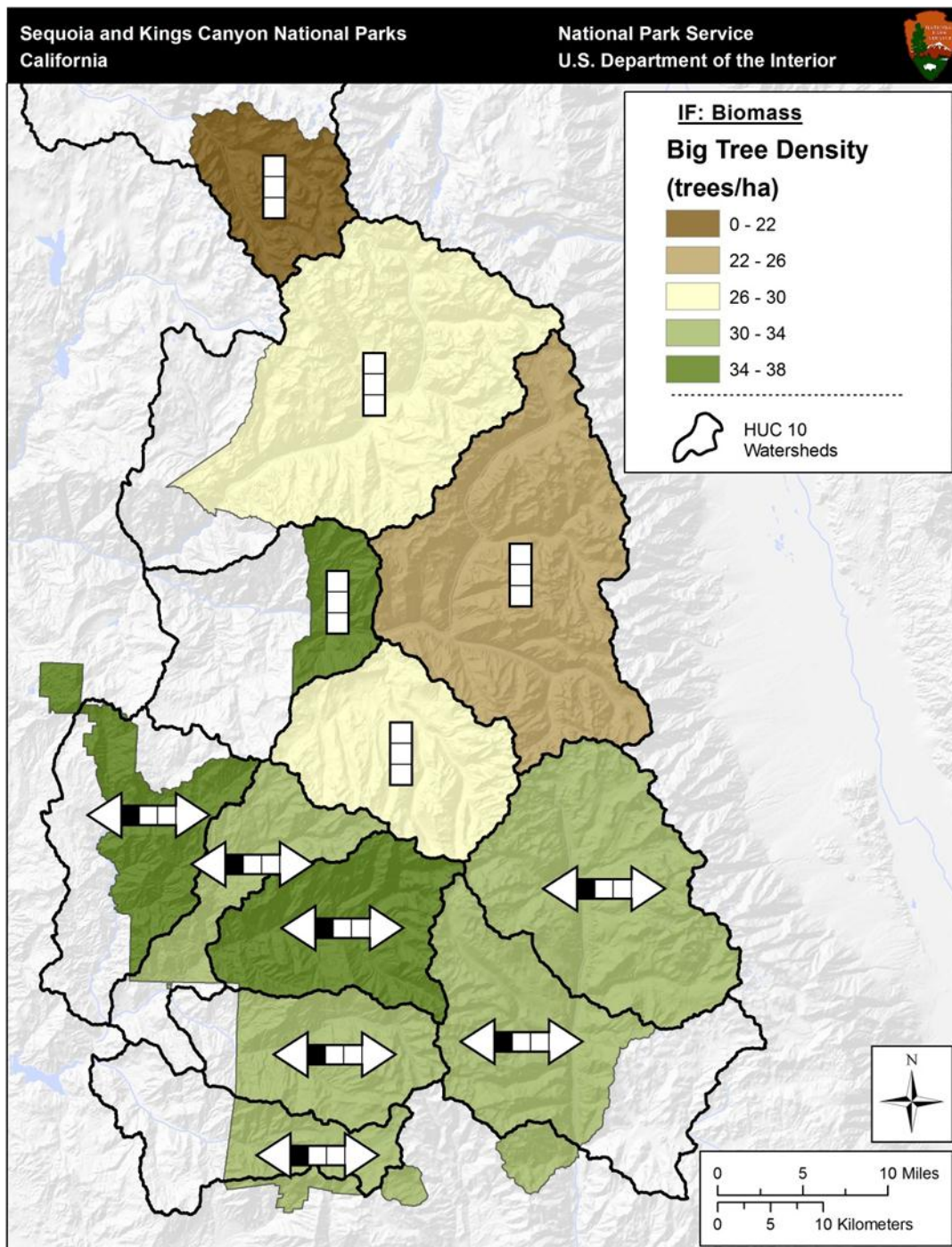
The asymmetric diameter-class distributions (Fig. 2) used as the references for the departure index generated corresponding asymmetric indices. The strong right-skew created indices that typically varied from -0.5 to 1.5. Positive values indicate more trees in the larger size classes and fewer in the smaller size classes. The maximum value (1.5) would mean that all the trees were in the largest diameter class. Negative values indicate more trees in smaller size classes and fewer in the larger size classes. The minimum value (-0.5) would mean that all the trees were in the smallest diameter class. We considered shifts in either direction potential cause for concern and thus rated changes in the departure index of $\pm 10\%$ as caution and changes $> \pm 25\%$ as concern (Table 4).

In summary, our criteria for ecological integrity (Table 4) were based on three separate sources. Average patch size and snag abundance followed the logic developed by Tierney et al. (2009). Big tree density (with big trees defined by Lutz et al. 2009) and aboveground live tree biomass in SEKI was compared to similar forest types in the surrounding area as defined by the PACE. Finally size distribution and largest patch index were “self-referential” in that intact forests across the HUC 10 watersheds were ranked relative to all intact forests in the Parks.

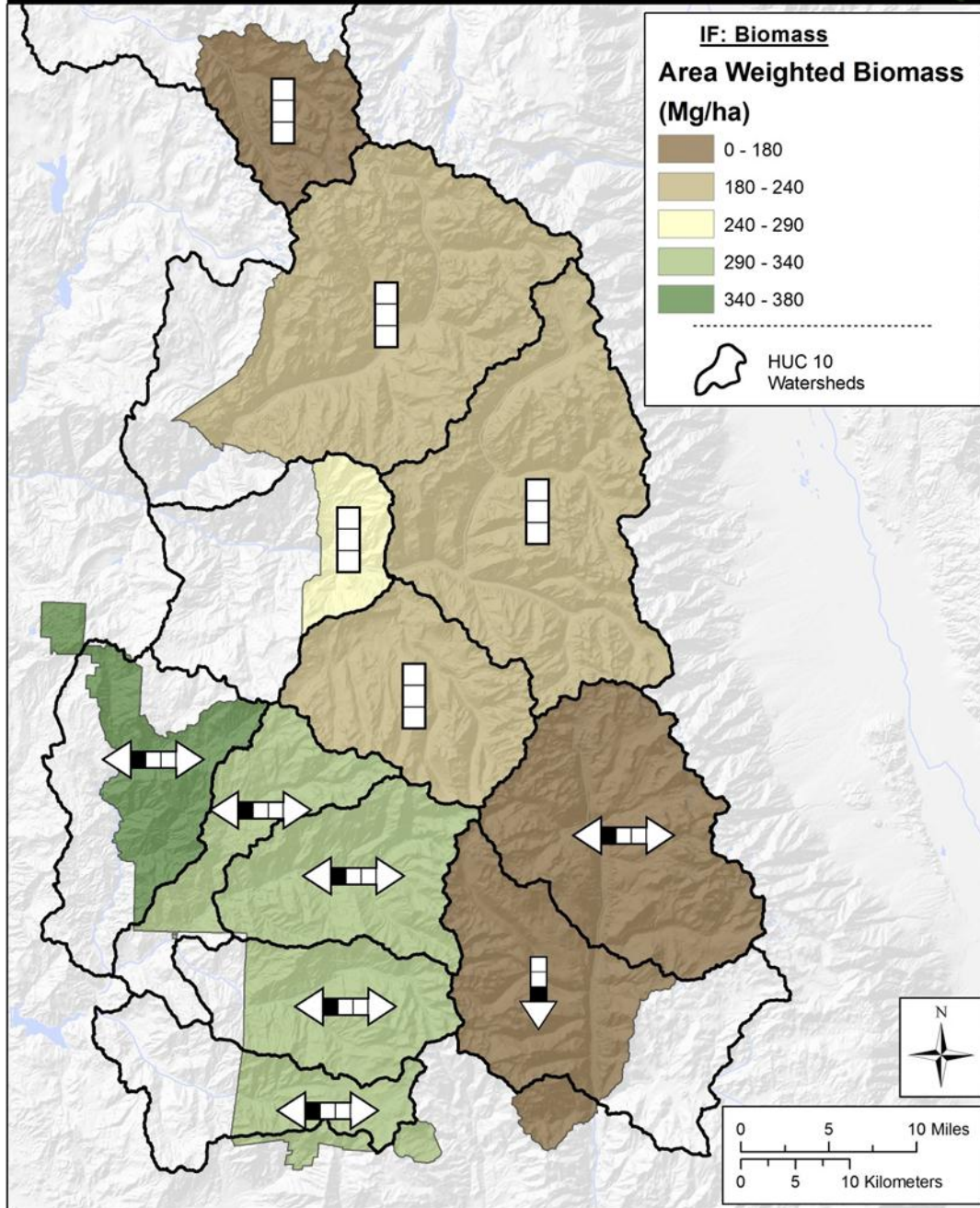
These metrics are admittedly compromises based on available knowledge and information. Thus their limitations must be acknowledged. For example, patch size as an indicator of integrity can be deceiving. Severe fires outside of the normal range of variation can create large contiguous patches of post-fire vegetation (i.e. shrub fields) that are uncharacteristic of the intact forest. Also too many snags in a forest can be indicative of a decline (chronic moisture deficit) or a recent disaster (e.g., severe fire, pest outbreak) rather than integrity. Finally live tree biomass in SEKI varies by forest composition (Table 1). Any variation in the relative abundance of the different forest types in the HUC 10 units may confound the interpretation of lower biomass. Despite their weaknesses, these standards are based on insights presented in the peer-reviewed literature and/or results calculated from relevant data. They provide a reasonable starting point and will only improve as better data and more knowledge is brought to bear on the question of quantifying forest integrity.

Results/Discussion

At the scale of the watershed, the SEKI intact forests had high ecological integrity. For most measures and most watersheds (Table 5), the results from the NRI data exceeded the standards for a good rating (Table 4). In terms of landscape structure, large patch sizes predominated in a contiguous matrix of forests. There were a few watersheds where the largest patch index was less than 11% with the low value of 7%, which was observed in the Lower South Fork Kings River watershed. In terms of forest structure, the density of big trees far exceeded comparable forests in the PACE (Map 5, Table 2) with the median big tree density more than double the standard (Table 4). There was also a sufficient supply of snags across the landscape with only the Golden Trout Creek watershed registering a low percentage of snags (Table 5). In terms of forest function, there was very little deviation in tree size class distributions from the reference condition. None moved more than 10% in either direction (Table 5). In a manner similar to big tree density, live tree biomass in the SEKI intact forest (Map 6) was much greater on average than the biomass in the PACE (Table 3). The one location for caution was the Golden Trout Creek. Not only was its average biomass, 126 Mg ha^{-1} , the lowest observed, but it was also a watershed with a low percentage of snags (Table 5).



Map 5. Spatial and temporal trends in big tree density for Sequoia Kings Canyon National Parks. Confidence assessment in legend refers to temporal trend. Data source: NRI and Stephenson vegetation plots.



Map 6. Spatial and temporal trends in aboveground live tree biomass for Sequoia Kings Canyon National Parks. Confidence assessment in legend refers to temporal trend. Data source: NRI and Stephenson vegetation plots.

Temporal Analyses

Introduction

We assessed trends in three of the ecological integrity measures – aboveground live tree biomass, big tree density, and size distribution. We used data from Stephenson’s forest gradient study (Stephenson 1998). Stephenson collected tree data for intact forests across the Kaweah River and Kern River in SEKI. His sampling occurred approximately 5 years prior to the NRI sampling. Most of Stephenson’s data were collected in 1983 (N. Stephenson, USGS WERC, personal communication). The NRI data used in this analysis was collected over 12 years with the first plot sampled in 1985 and the last plot sampled in 1996 but the modal and mean sample year for the NRI plots was 1988. Note that Stephenson (1998) did not record snags thus we cannot calculate snag density. We also cannot check for trends in the landscape structure since these metrics were derived from the SEKI vegetation map.

Methods

We calculated tree biomass, big tree density and size distribution as described above. However we needed to convert Stephenson’s categorical dbh measurements for the trees to exact estimates. Trees less than 140 cm in dbh were placed into 5-cm dbh classes. Trees larger than 140-cm dbh were measured. Since the dbh distributions followed an exponential decline, we assigned the geometric mean value to every tree in a class. For example, trees in the 5-10 cm dbh class were assigned an exact dbh value of 7.36. In effect the geometric mean produced only a slightly smaller estimate than assuming an arithmetic mean. To quantify shifts in the size distribution, we calculated the departure index for each plot using the observed mean distribution for the NRI plots from the same forest alliance as reference. Thus a departure index < 0 implies a greater proportion of smaller trees relative to the NRI data, an index > 0 implies a greater proportion of larger trees, and an index $= 0$ implies no difference.

Results from the Stephenson plots were first summarized by intact forest alliance and then interpolated to the level of the watershed as described for the NRI data. In short, the results for each watershed were an alliance-weighted average with the values drawn from the Stephenson data. We then looked for differences between the two estimates to detect any trends. For big tree density and live tree biomass, we considered differences that exceeded the NRI-base mean $\pm$ one standard deviation as evidence for a trend. For the departure index, which did not follow a normal distribution, we considered watershed values that moved 25% in either direction as indication of a trend.

Results/Discussion

Estimates of aboveground live tree biomass and big tree density were generally greater for the Stephenson plots (Table 6). For some alliances, the Stephenson data produced estimates of live tree biomass that were two to three times more than the NRI values (e.g., ponderosa pine forest and giant sequoia alliances, Table 6). However alliances with large differences in means were accompanied by high variance. For example, the percent error (percent error = standard error/mean) in the mean live biomass estimate for the ponderosa pine forest alliances was 90%; for the giant sequoia alliance, the percent error was 49%. Certainly the small number of Stephenson plots in some alliances contributed to these high estimates of variation. For other alliances, the means were more comparable. In particular the estimates were similar for two

common forest types -- lodgepole pine and white fir-sugar pine. Together these two alliances accounted for 37% of the intact forest.

Tree-size distributions calculated from the Stephenson data followed the pattern observed for the NRI plots. For all alliances, tree abundance declined exponentially from the smallest to largest diameter class (Fig. 2B). We also detected no systematic shifts in the relative abundance of big trees for most forest alliances (Fig. 4). The exceptions included the lodgepole pine (Fig. 4B) and western white pine forest alliance (Fig. 4H) where there seemed to be a greater proportion of trees in the larger size classes in the Stephenson plots (departure index > 0).

At the watershed level, estimates of big tree density and aboveground live tree biomass based on the Stephenson data were greater than the averages derived from the NRI data. These differences suggested a general trend toward declining big tree density and live tree biomass. However only in one instance - live tree biomass in the Golden Trout Creek watershed (Kern River drainage) - was the difference greater than one standard deviation beyond the mean (Map 6). In terms of tree size distribution, there was no meaningful trend observed in the departure indices for any watershed.

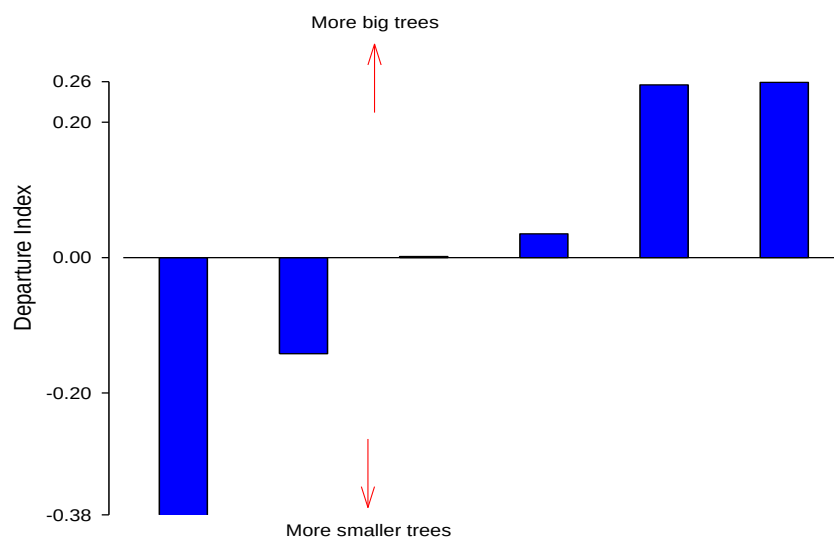
Table 6. Comparison of the composition and structure of the major forest alliances in Sequoia-Kings Canyon National Parks. Results based on National Resource Inventory (NRI) data (Graber and Haultain 1993) and Stephenson forest gradient data (Stephenson 1998). Means with standard errors in parentheses reported for all metrics. The sample size (n) represents the number of plots in the alliance with first number for NRI data and the second for Stephenson data.

Alliance	NRI				Stephenson			
	Density (stems ha ⁻¹)	Big Tree Density (stems ha ⁻¹)	Basal Area (m ² ha ⁻¹)	Aboveground Biomass (Mg ha ⁻¹)	Density (stems ha ⁻¹)	Big Tree Density (stems ha ⁻¹)	Basal Area (m ² ha ⁻¹)	Aboveground Biomass (Mg ha ⁻¹)
Quaking Aspen (n=3,0)	112 (111)	0	1.5 (1.8)	3.3 (4.1)	--	--	--	--
Black oak (n=5,6)	590 (71)	40 (12)	41.4 (15.2)	250.7 (86.0)	1203 (484)	86 (22)	76.6 (21.5)	491.6 (171.8)
Lodgepole pine (n=48,20)	467 (70)	31 (5)	32.3 (3.7)	177.5 (27.0)	387 (78)	48 (10)	41.8 (5.2)	201.4 (30.2)
Ponderosa pine Woodland (1,0)	170	10	24.6	116.0	--	--	--	--
Ponderosa pine Forest (n=13,4)	777 (164)	42 (8)	46.5 (9.0)	254.0 (52.5)	775 (389)	47 (17)	81.1 (58.9)	819.7 (733.3)
Giant sequoia (n=21,11)	700 (176)	46 (6)	101.1 (18.4)	895.5 (220.9)	748 (259)	58 (9)	211.8 (83.6)	2,268.4 (1113.6)
Red fir (n=20,17)	461 (67)	30 (6)	50.2 (5.3)	274.4 (32.9)	532 (89)	54 (9)	81.8 (10.9)	477.8 (81.3)

Table 6. Comparison of the composition and structure of the major forest alliances in Sequoia-Kings Canyon National Parks. Results based on National Resource Inventory (NRI) data (Graber and Haultain 1993) and Stephenson forest gradient data (Stephenson 1998). Means with standard errors in parentheses reported for all metrics. The sample size (n) represents the number of plots in the alliance with first number for NRI data and the second for Stephenson data (continued).

Alliance	NRI				Stephenson			
	Density (stems ha ⁻¹)	Big Tree Density (stems ha ⁻¹)	Basal Area (m ² ha ⁻¹)	Abovegro und Biomass (Mg ha ⁻¹)	Density (stems ha ⁻¹)	Big Tree Density (stems ha ⁻¹)	Basal Area (m ² ha ⁻¹)	Aboveground Biomass (Mg ha ⁻¹)
Red fir-White fir (n=22,15)	561 (121)	35 (5)	44.2 (6.1)	281.4 (47.6)	477 (124)	43 (7)	58.8 (10.3)	414.5 (101.0)
White fir-Sugar pine (n=63,14)	631 (64)	34 (3)	51.6 (4.0)	338.2 (30.8)	918 (179)	37 (10)	54.5 (8.7)	313.8 (56.2)
Western white pine-Lodgepole (n=8,7)	321 (51)	15 (6)	21.0 (4.6)	93.5 (22.7)	214 (85)	26 (14)	39.7 (16.1)	240.3 (104.1)

A. Black Oak Forest Alliance



B. Lodgepole Pine Forest Alliance

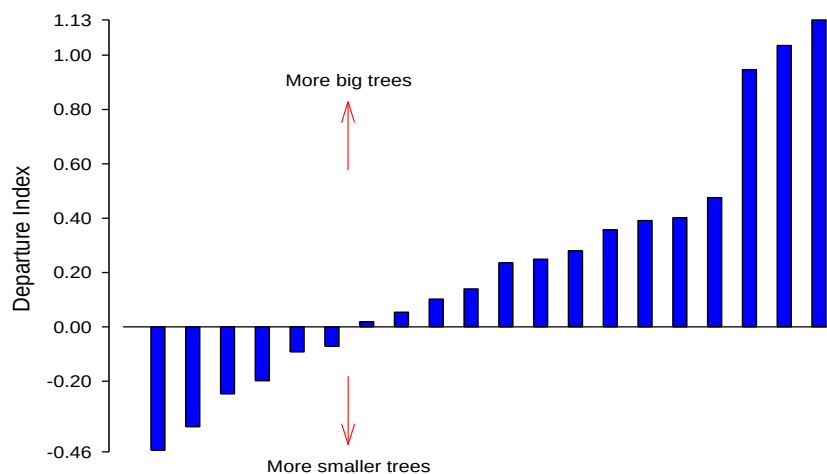
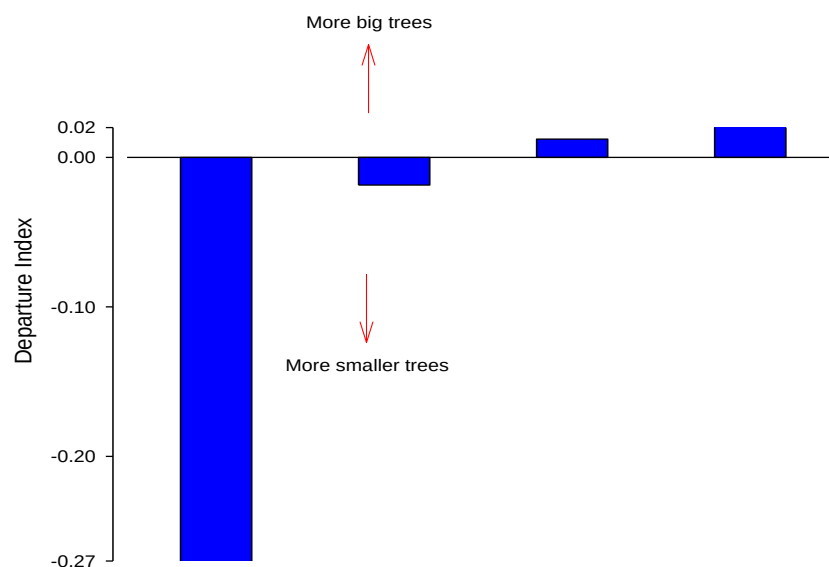


Figure 4.A-B. Differences in size distribution between Stephenson Gradient Analysis and Natural Resource Inventory at Sequoia – Kings Canyon National Parks for A) black oak forest alliance and B) lodgepole pine forest alliance. Change measured using the Departure Index for major alliances in the intact forest. Bars (distributed along x-axis) represent the Departure Index for each plot in the Stephenson data compared to the NRI reference.

C. Ponderosa Pine Forest Alliance



D. Giant Sequoia Forest Alliance

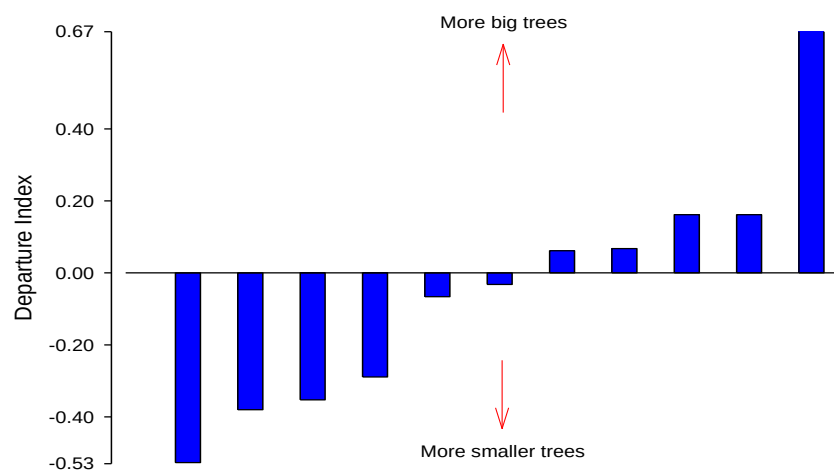
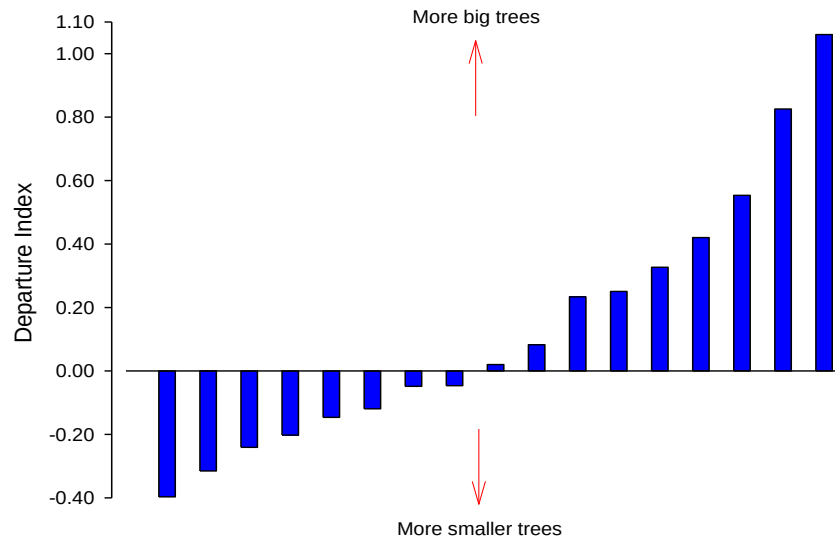


Figure 4.C-D. Differences in size distribution between Stephenson Gradient Analysis and Natural Resource Inventory at Sequoia – Kings Canyon National Parks for C) ponderosa pine forest alliance and B) giant sequoia forest alliance. Change measured using the Departure Index for major alliances in the intact forest Bars (distributed along x-axis) represent the Departure Index for each plot in the Stephenson data compared to the NRI reference.

E. Red Fir Forest Alliance



F. Red Fir - White Fir Forest Alliance

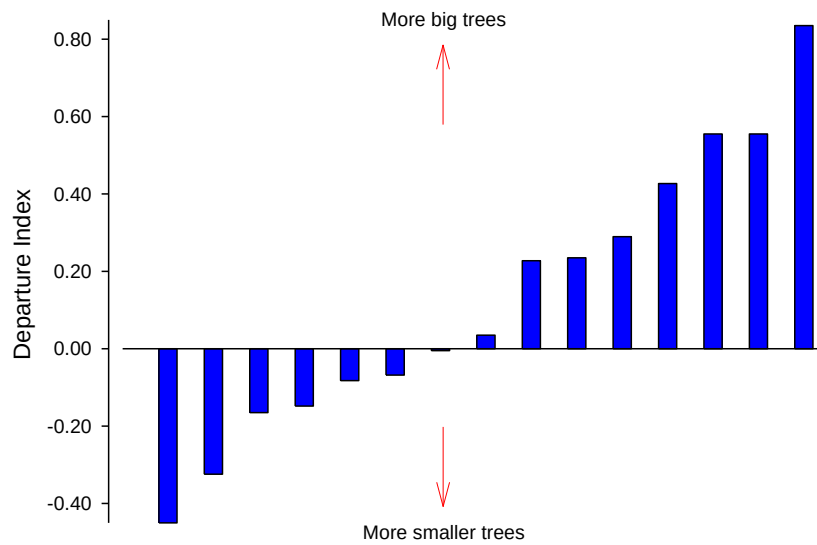
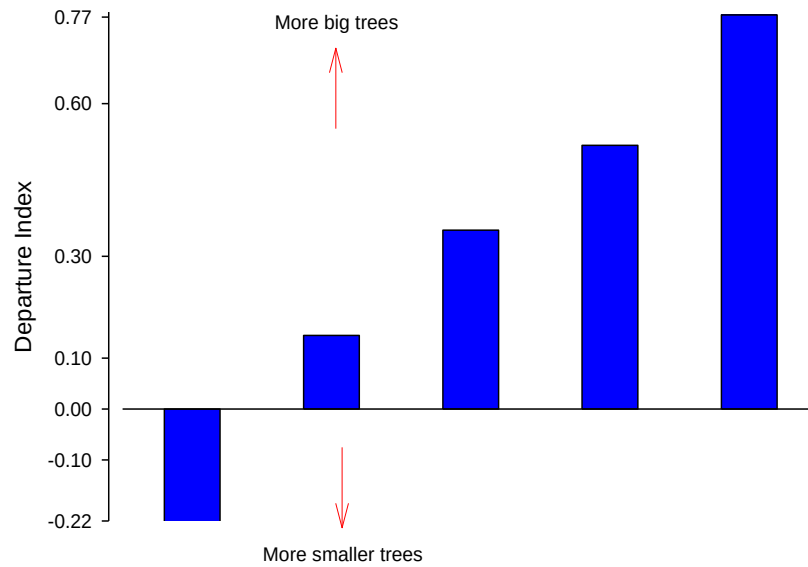


Figure 4.E-F. Differences in size distribution between Stephenson Gradient Analysis and Natural Resource Inventory at Sequoia – Kings Canyon National Parks for E) red fir forest alliance and F) reid fir – white fir forest alliance. Change measured using the Departure Index for major alliances in the intact forest Bars (distributed along x-axis) represent the Departure Index for each plot in the Stephenson data compared to the NRI reference.

H. Western White Pine Forest Alliance



G. White Fir - Sugar Pine Forest Alliance

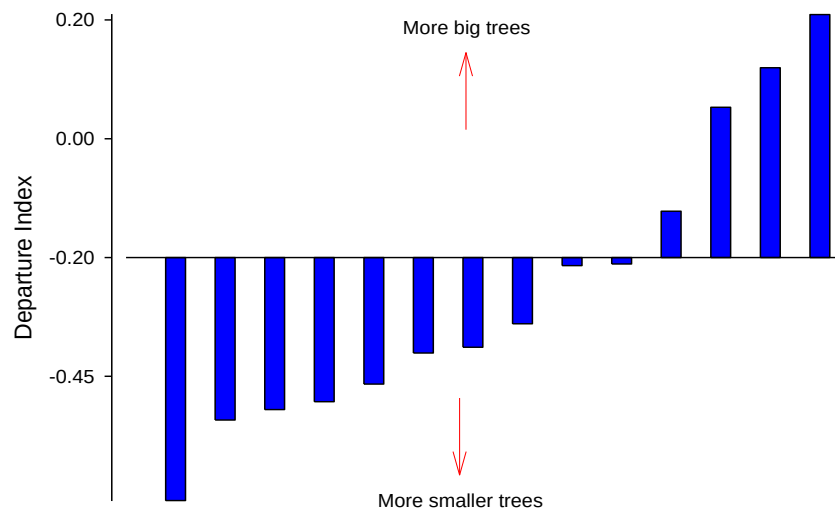


Figure 4.G-H. Differences in size distribution between Stephenson Gradient Analysis and Natural Resource Inventory at Sequoia – Kings Canyon National Parks for G) white fir – sugar pine forest alliance and H) western white pine forest alliance. Change measured using the Departure Index for major alliances in the intact forest. Bars (distributed along x-axis) represent the Departure Index for each plot in the Stephenson data compared to the NRI reference.

Analysis Uncertainty

Our assessment of ecological integrity was data-driven. Calculations were based on either data-derived standards (live tree biomass, snag density) or peer-reviewed approaches (area-weighted patch size, largest patch index, big tree density, and departure index). We relied on basic measures of central tendency and variation to summarize results by intact forest alliance. For metrics that followed a normal distribution, we used means and standard deviations. For metrics that followed asymmetric distributions, we reported the median and the interquartile range. These analyses do not introduce uncertainty in their own right. They do inherit the measurement and sampling errors from the vegetation plots. In addition, our projections of results to the HUC 10 watershed imply a spatial specificity that is not supported by the plot network in all cases. We discuss these limitations below (**Section: Levels of confidence in assessment**).

Stressors

Altered fire regime

A suite of abiotic and biotic stressors pose risks to the integrity of the intact forests at Sequoia-Kings Canyon National Parks (Exline et al. 2009). Foremost among these is the altered fire regime. Fire plays a fundamental role in the ecology of the intact forest alliances (Caprio and Lineback 2002, Fites-Kaufman et al. 2007). Disruption of the pre-European fire regime in Sequoia-Kings Canyon National Parks since 1860 has created the potential for large, stand-destroying fires in forests where such catastrophic events happened rarely if ever in the past (Caprio and Graber 2000). To measure the extent of the alteration to the fire regime and to monitor management actions, Caprio et al. (1997) developed a geospatial model that estimated the extent of departure from the historic fire return interval for each vegetation type in the Parks (Fire Return Interval Departure, FRID). FRID values are calculated as the ratio of the time since last fire to the historic fire return interval. In SEKI, the indices ranged from -1 (just burned area) up to 16 which indicates that the time since last fire is 16 times longer than the historic fire return interval. See Caprio et al (1997) for a complete discussion of the index.

We used the 2010 FRID database maintained by the Parks to quantify the stress imposed by the altered fire regime. Specifically, we calculated the means of the raw FRID values for the intact forest alliances (Table 7) and for the intact forests in each HUC 10 watershed (Table 8). Note that since the FRID is effectively normalized by vegetation type (Caprio et al. 1997), area-weighted means do provide unbiased summaries for defined landscape units and allow direct comparison among alliances and watersheds. We adapted the rating system of Caprio et al. (1997) to describe the extent of the risk posed by departures from the historic fire regime. Average FRID values < 0 indicate that the historic fire return interval has been maintained (Caprio et al. 1997) and thus forests in this condition were ranked as “good.” FRID indices ≥ 0 and ≤ 2 indicate that the time since the last fire is up to two times longer than the maximum return interval. We assigned FRID indices in this range as reason for “caution”. If the mean FRID > 2 , we considered the altered fire regime as a cause for “concern.”

For more than 40% of the intact forest, fire has been absent for two or more cycles placing much of the intact forest at risk of severe fire (Table 7). Among the 12 alliances, giant sequoia had the most extreme index (FRID = 6.2). Given that the estimated maximum fire return interval for this

alliance is approximately 16 years (Caprio and Lineback 2002); much of the giant sequoia forest has not burned in the last 100 years. Spatially, nearly the entire Kawaeh River drainage (Map 7, Table 8) had average FRID values > 2. The concentration of giant sequoia alliances in the Kawaeh (7% of total area of intact forest, Map 1, Map 4) contributed to the observed departure in the Kawaeh watersheds. The intact forests located in the other SEKI watersheds were much closer to the historic fire regime (Map 7).

Table 7. Summary of the magnitude of abiotic stressors by each intact forest alliance at Sequoia-Kings Canyon National Parks. Values are the means for all polygons in each intact forest alliance.

Alliance	Total Area (ha)	Ozone 8-hr standard (ppb)	Nitrogen Deposition (kg ha ⁻¹ yr ⁻¹)	Fire Return Interval Departure	Water Deficit 30-yr mean (mm)
Quaking aspen (2010)	2,492	68.8	2.1	-0.3	133
Black oak (2020)	5,040	86.5	5.6	2.3	402
Lodgepole pine (3020)	20,478	71.4	2.1	-0.5	141
Ponderosa pine (woodland, 3050)	198	89.2	5.4	-0.2	388
Ponderosa pine (forest, 3060)	5,539	86.0	4.8	-0.4	398
Giant sequoia (4020)	4,613	90.9	6.6	6.2	386
Mountain hemlock (4030)	561	71.8	2.0	2.5	67
Red fir (4050)	19,973	80.3	2.5	2.9	197
Red fir-White fir (4070)	12,967	83.8	3.2	-0.8	255
White fir-Sugar pine (4080)	16,062	86.7	5.1	0.7	362
Incense cedar (4111)	132	78.8	4.0	2.6	431
Western white pine- Lodgepole (4540)	10,252	75.6	2.2	2.9	172

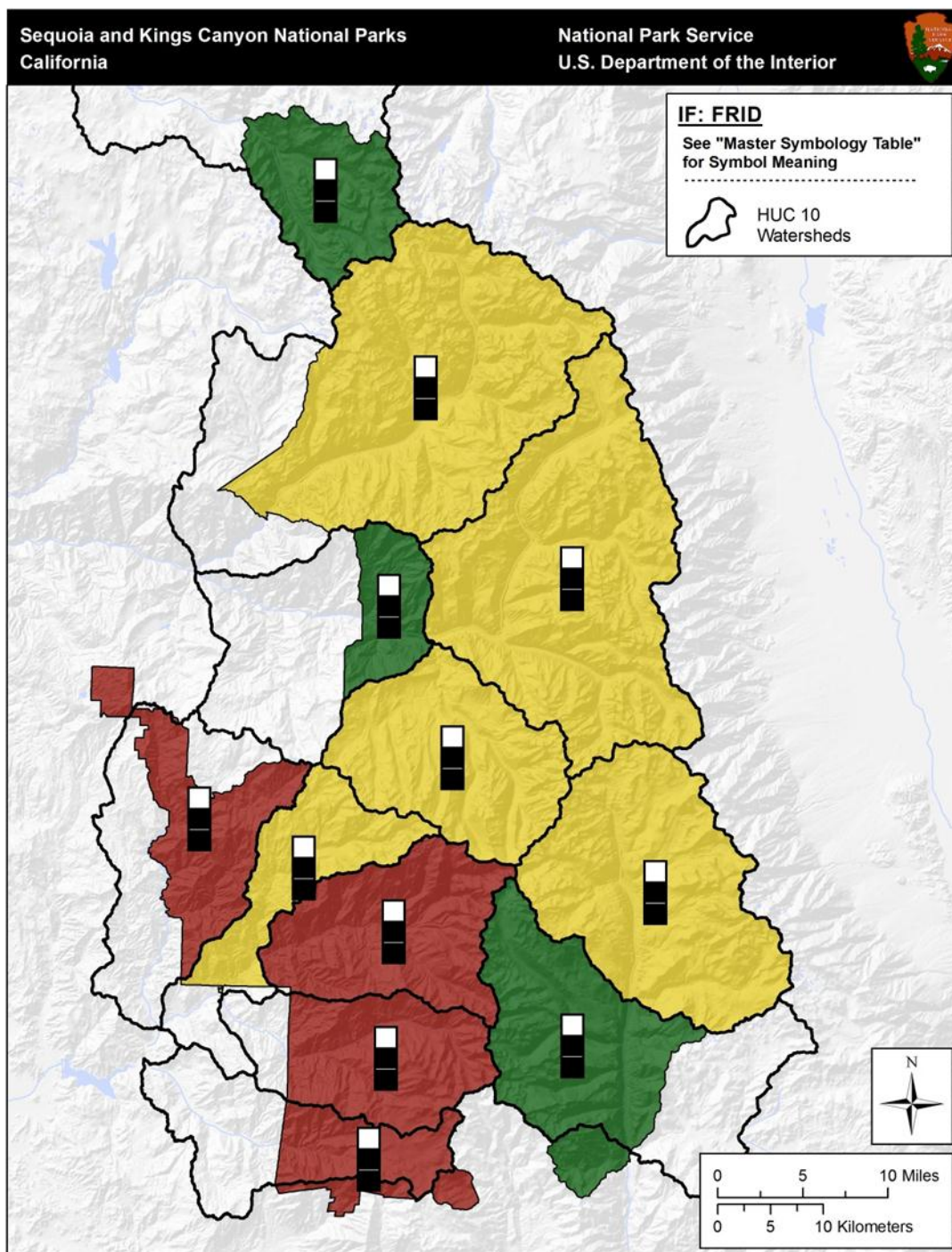
Ozone pollution

The interaction of geography, climate, and emissions patterns shape ozone production and transport, contributing to high ozone exposure in the southern Sierra, particularly during California's warm, dry, summers. Pollutants emitted in the San Francisco Bay Area and Central

Valley are forced south and east by prevailing winds. Once deep in the San Joaquin Valley, the Sierra Nevada and Tehachapi mountains block passage of air, resulting in the production and accumulation of ozone in the southern Sierra (Carroll et al. 2003, Beaver and Palazoglu 2009). Ambient ozone levels typically peak in late summer, at the nadir of water availability for perennial plants (Grulke 2002).

Table 8. Distribution of the major abiotic stressors of the intact forest at Sequoia-Kings Canyon National Park. Results pertain only to the intact forests in each HUC 10 watershed. Cell color codes reflect the criteria discussed in the text. Coding: green = “good”; yellow= “caution”; red = “concern.” No shading indicates no criterion for measuring stress.

HUC 10	Ozone concentration 8-hr standard (ppb)	Nitrogen deposition (kg ha ⁻¹ yr ⁻¹)	Fire return departure index (-0.9,13.4)	Water deficit 30-yr mean (mm)
Middle Fork Kings River	76.5	2.0	0.4	151
Upper South Fork San Joaquin River	66.9	2.0	-0.4	24
East Fork Kaweah River	84.4	3.5	2.1	283
North Fork Kaweah River	92.2	8.6	2.9	465
Marble Fork Kaweah River	87.7	4.8	1.7	318
Lower South Fork Kings River	81.2	1.7	-0.1	193
South Fork Kaweah River	87.7	4.8	2.2	336
Golden Trout Creek- Kern River	70.3	1.7	-0.2	238
Upper South Fork Kings River	70.7	2.1	0.2	128
Rock Creek-Kern River	63.8	1.5	0.1	155
Middle Fork Kaweah River	83.1	2.7	2.6	264
Roaring River	76.7	2.0	0.0	176



Map 7. Condition assessment of the fire return interval departure index for intact forests in Sequoia Kings Canyon National Parks. Data source: SEKI GIS databases. See Master Symbology Table in Appendix 3.

Southern Sierra ozone concentrations regularly surpass national ambient air quality standards (NAAQS) with summer average daily maximums of 70-95 parts per billion by volume (Carroll et al. 2003). Sequoia and Kings Canyon National Parks are perennially among the National Parks with the worst air quality; the Park Service describes the effects of ozone here as “among the most severe in North America for non-point source pollution” (Sullivan et al. 2001, Landers et al 2008). Sequoia and Kings Canyon National Parks regularly exceeded the NAAQ 8-hour standard (75 ppb) during the summer months: 66 days were in exceedance at Ash Mountain in 2010 (NPS 2010b).

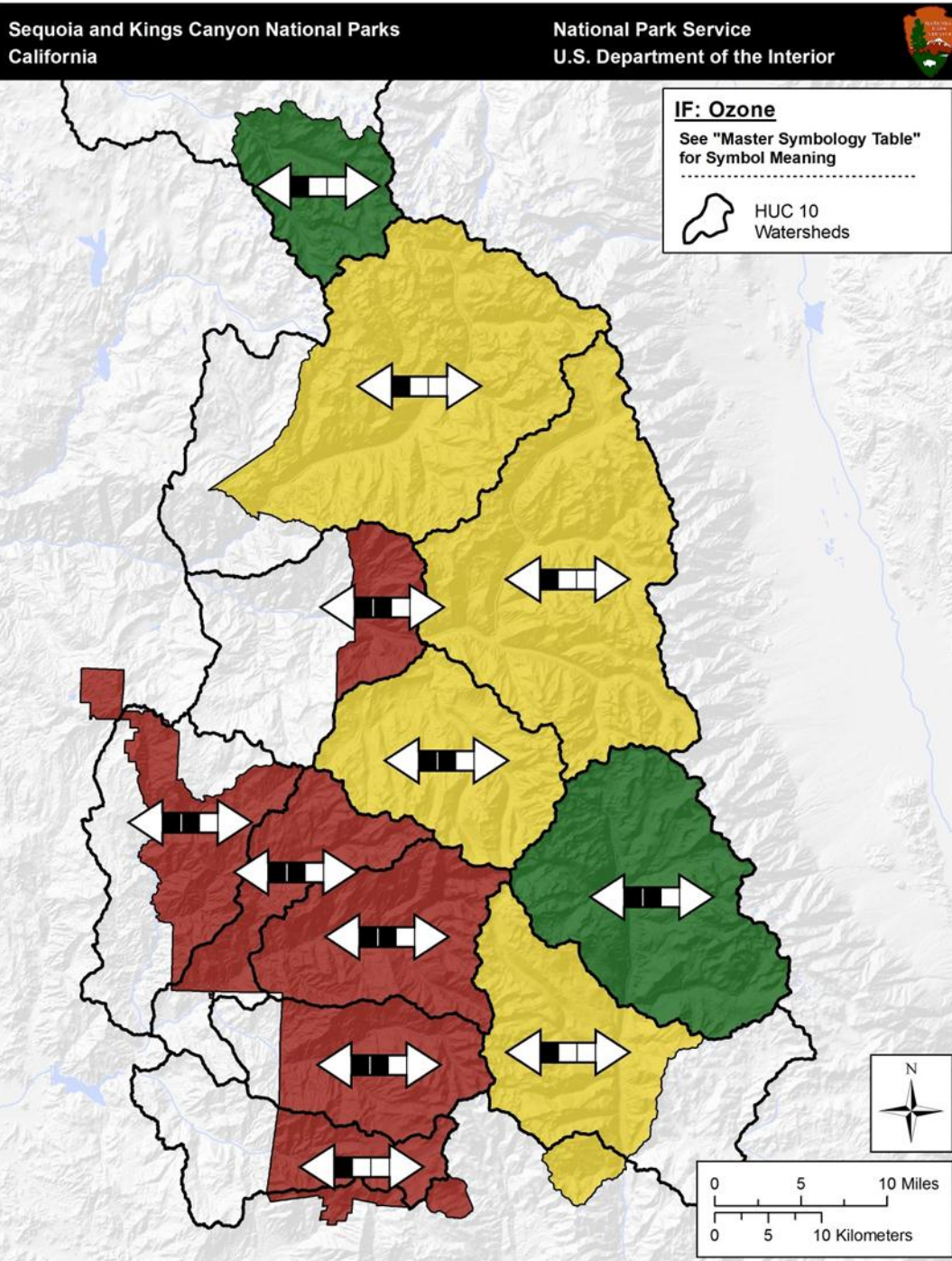
Ozone levels at or above the NAAQ standard can cause damage in sensitive species. For example, ozone injury to ponderosa and Jeffery pines is evidenced by yellowing of needles in a distinctive pattern known as chlorotic mottle. Other bioindicators include an increase in deciduous needle habit, shortened needles, fewer annual whorls per branch, and decreased crown density (Grulke 1998). Bytnerowicz et al. (1999) documented extensive ozone damage in the southern Sierra; Duriscoe (1990) recorded specific symptoms of ozone damage to sensitive pine species in SEKI intact forests in the western edge of the Parks. Importantly, Panek et al. (Air Quality NRCA) demonstrated that the index of ozone damage on three-needled pines recorded by Duriscoe (1990) from field surveys corresponded to mean ozone concentration. Thus we used the ozone condition assessment developed by Panek et al. (Air Quality NRCA) to quantify the potential impact of ozone on the intact forests (Table 8). Specifically, “concern” was defined as ozone exposure greater than the 80 ppb (the federal 8H primary standard threshold of 75 ppb + 5 ppb to account for uncertainty). “Good” was 8hr ozone concentrations < 70 ppb. “Caution” was assigned to concentrations between these values (70-80 ppb). For the intact forests in each HUC 10 watershed, we calculated the average 8hr ozone exposure. Trend and confidence in the assessment were inherited from the Panek et al. (Air Quality NRCA) values.

Ozone exposure to the intact forests in the Kaweah drainage exceeded the 8-hr federal standard (Map 8, Table 8). These levels lead to observable injury in sensitive species such as Jeffrey pine and ponderosa pine (Panek et al. Air Quality NRCA). Outside of the Kaweah, most of the intact forest was exposed to moderate levels of ozone. The giant sequoia forest alliance experienced the highest ozone pollution with the average 8-hr exposure in excess of 90 ppb (Table 7).

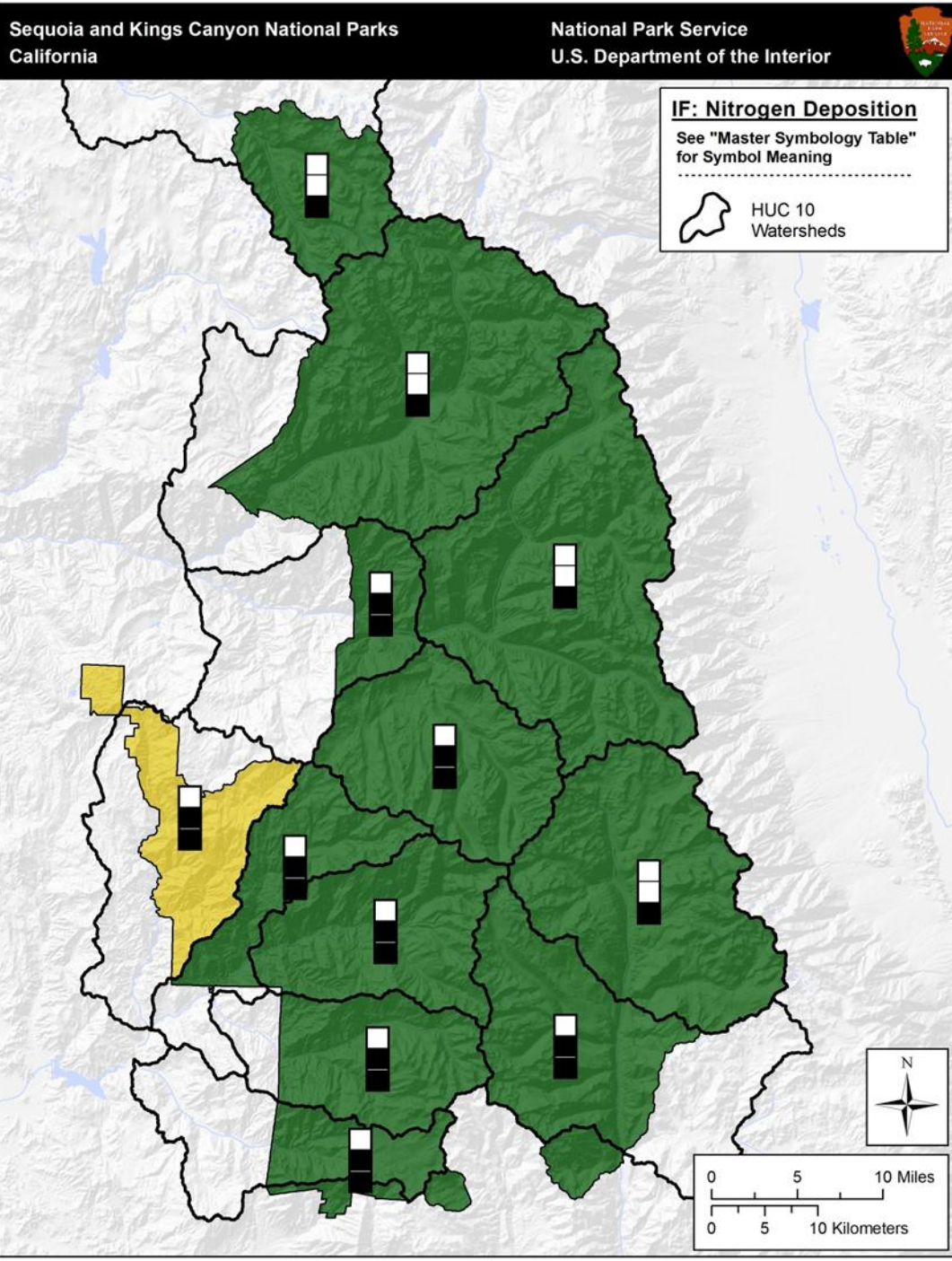
Nitrogen pollution

We used the critical loads for nitrogen for the mixed conifer forest as defined by Fenn et al. (2010) to assess the stress imposed on the intact forest by nitrogen pollution. As for ozone, we used the projections developed by Panek et al. (Air Quality NRCA) and summarized exposure for the intact forests in each HUC 10.

Across the Parks, nitrogen deposition was generally below the threshold considered damaging to ecosystems (< 5.2 kg ha⁻¹ yr⁻¹). Only the ponderosa pine woodland and the giant sequoia alliances received nitrogen in the moderate range (Table 7). Only the North Fork Kaweah watershed (Map 9) received levels of nitrogen above the threshold.



Map 8. Condition assessment of ozone exposure for intact forests in Sequoia Kings Canyon National Parks. Data source: NRCA Air Quality Assessment (Panek, this report). See Master Symbolology Table in Appendix 3.



Map 9. Condition assessment of nitrogen deposition for intact forests in Sequoia Kings Canyon National Parks. Data source: NRCA Air Quality Assessment (Panek, this report). See Master Symbology Table in Appendix 3.

Exotic pests

Pests refer to both insects and diseases that prey on trees. Their activity is a natural aspect of forest ecosystems. For example, more than half (54%) of 3,743 tree mortalities observed from 1982-2005 in the 14 forest demography plots in SEKI and YOSE (van Mantgem and Stephenson 2007) noted the presence of a forest pest (Das et al. 2011). Most of these pests are native bark beetles, wood borers, defoliators and diseases. However throughout California's forests, the ratio of exotic (non-native) to native pests has been increasing over time (FRAP 2010). In addition the population density, and consequently the impact, of some native organisms (e.g., bark beetles) have increased in forests simultaneously stressed by drought and high tree density (Christensen et al. 2008). Models of bark beetle population dynamics based on several climate warming scenarios document an increased risk of outbreaks and widespread tree mortality in western conifer forests (Bentz et al. 2010). Perhaps at the most immediate risk to native bark beetles, are stands dominated by lodgepole pine. Drought and other stressors have triggered widespread eruptions of mountain pine beetle (*Dendroctonus ponderosae*) in lodgepole pine populations throughout western North America (Raffa et al. 2008). Forests with a significant component of lodgepole pine cover more than 31,000 ha in SEKI (31% of the intact forest, Table 1). To date, we have not observed an eruption like those reported for British Columbia and the Rocky Mountains (Raffa et al. 2008). But, mountain pine beetle is present in the Parks (A. Das, USGS WERC, personal communication). Given the trends in climate, the potential for outbreaks exists (Evangelista et al. 2011). Despite the threats posed by newly introduced pests and by increased virulence of native pests, it is an "old" introduced disease – white pine blister rust – that poses a clear and present danger to the intact forests at SEKI.

White pine blister rust (*Cronartium ribicola*) ranks as one of the most destructive disease introductions in history (Tomback and Achuff 2010). Its life cycle is complex with five different spore stages on two completely unrelated hosts: five-needled pines (including sugar pine and western white pine), and shrubs in the genus *Ribes* (gooseberries and currants). These shrubs are ubiquitous in the understories of the Sierran conifer forests. A recent discovery found that plants in the genera *Castilleja* and *Pedicularis* can also serve as alternate hosts for *C. ribicola* (McDonald et al. 2006). The spores that infect pines are produced in the late summer/early fall under cool and very moist conditions. This cool and moist environment must be maintained for the spores to survive and infect the pine host via the leaf stomata. The infection spreads from the leaf to the branch or bole. The resulting cankers can kill the tree directly by girdling it or indirectly by predisposing an infected tree to secondary pests and pathogens (Geils et al. 2010).

Measures to eliminate or contain this pest constitute the most extensive and expensive effort of its kind in US forestry (Maloy 1997). In general, the blister rust control program is regarded as a failure. Initially there was hope that the epidemic could be constrained to a few outbreak centers in the Sierra Nevada. The dry Mediterranean summer was considered an environmental barrier to the dispersal of the spores that in combination with the vigilant removal of afflicted trees, could limit the spread. However "waves" of infection in the 1970s and 1980s distributed the disease over large areas of California. Although mortality due to the blister rust is still patchy throughout the Sierra Nevada, there is mounting evidence that where blister rust infections occur, declines in pine populations will follow (Kinloch and Dulitz 1990, van Mantgem et al. 2004). A single dominant gene exists in sugar pine that confers virtual immunity to blister rust (Kinloch et al. 1970), although the gene has low frequency in natural populations. Gene frequency varies

through the range of sugar pine from 0 in the southern Cascade Range to 0.08 in the southern Sierra Nevada (Kinloch 1992). At Mountain Home Demonstration Forest in the southern Sierra Nevada, Kinloch and Dulitz (1990) estimated a gene frequency of 0.076 in the seedling population. Thus approximately 15% of parent trees at Mountain Home have major gene resistance. At Blodgett Forest Research Station in the central Sierra Nevada, only 5% of the screened sugar pine parents (130 trees) were rust-resistant (R. Heald, UC Berkeley Center for Forestry, personal communication).

White pine blister rust was detected in SEKI in 1969 and has since spread throughout the range of the host species in the Parks (Duriscoe and Duriscoe 2002). Sugar pine is an important and widespread component of the intact forests at SEKI (Appendix 1, Map 10). On average, it accounts for more than 7% of the aboveground live tree biomass. Western white pine is much less abundant (<1% of the aboveground live tree biomass) but it is a dominant species in the western white pine-lodgepole pine forest alliance (Appendix 1). This alliance covers more than 100 km² of SEKI (Table 1, Map 1).

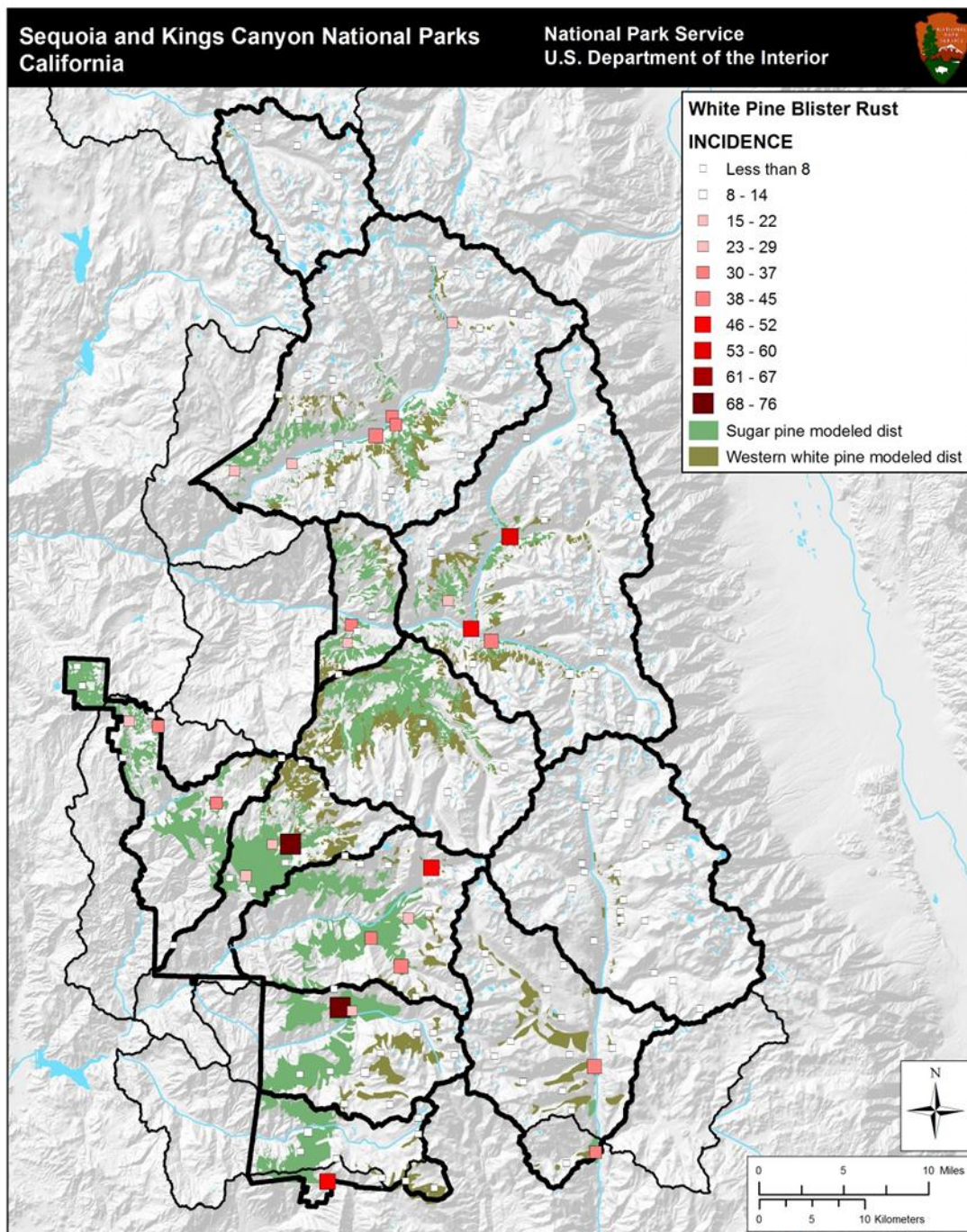
Almost 20% of the sugar pine trees in the blister rust monitoring plots had symptoms of the disease; for western white pine, it was slightly less than 5% (Duriscoe and Duriscoe 2002). However there was great variation in the level of infection. Many survey plots had no infected trees (Fig. 5). For example 12 out of the 43 survey plots with at least one sugar pine tree had no rust and 42 out of the 52 plots with at least one western white pine had no rust (data from Duriscoe and Duriscoe 2002). To summarize the distribution, we calculated the mean incidence of rust for each forest alliance (Table 9) and HUC 10 (Table 10). Incidence was calculated as the number of infected trees/total number of trees in the plots. As noted by Duriscoe and Duriscoe (2002), there is a strong elevational gradient in the level of infection with little evidence of rust at the highest elevation. For example, despite having 10 monitoring plots in the western white pine alliance, there was no sign of the rust (Table 9). In contrast, blister rust was common in alliances that occur at lower elevations (Table 9, Appendix 1).

Since rust is a non-native pest, we considered any presence as a stress. If the mean incidence was greater than 10%, we rated it cause for concern. In general, there was more rust in the sugar pine trees than western white pine and the incidence was highest in the Kaweah River drainage (Map 11). In the Marble Fork and Middle Fork of the Kaweah River, incidence for both species was more than 20%.

Drought

There is widespread concern that increased drought and heat-stress associated with climate change is leading to a worldwide increase in tree mortality (Allen et al. 2010). At SEKI and YOSE, the annual mortality rate of trees in the forest demography plots nearly doubled between 1983-2004 (van Mantgem and Stephenson 2007). Moreover, van Mantgem and Stephenson (2007) concluded that a significant increase in climatic water deficit during the same interval was the most likely explanation for the observed mortality rate. The phenomenon was consistently observed across the elevation gradient and major species groups in the intact forest. Thus we used water deficit as an index of drought stress. We calculated the climatic water deficit data (270-m resolution grids) averaged over the 30-year period from 1971 to 2000 (Rice and Bales, NRCA report) for the intact forest alliances and the HUC 10 watersheds. The 30-yr deficit varied

greatly by alliance (Table 7) and HUC 10 (Table 8). Such variation is expected given the individualistic response of species to climate and the strong topographic controls on climate (Whittaker 1975). Thus there is no absolute standard for quantifying drought stress at these scales. We presented these spatially interpolated values as a reference to compare the direction and magnitude of future changes.



Map 10. Incidence of white pine blister rust in relation to the modeled distributions of sugar pine (PILA10) and western white pine (PIMO10). Results from Duriscoe and Duriscoe (2002).

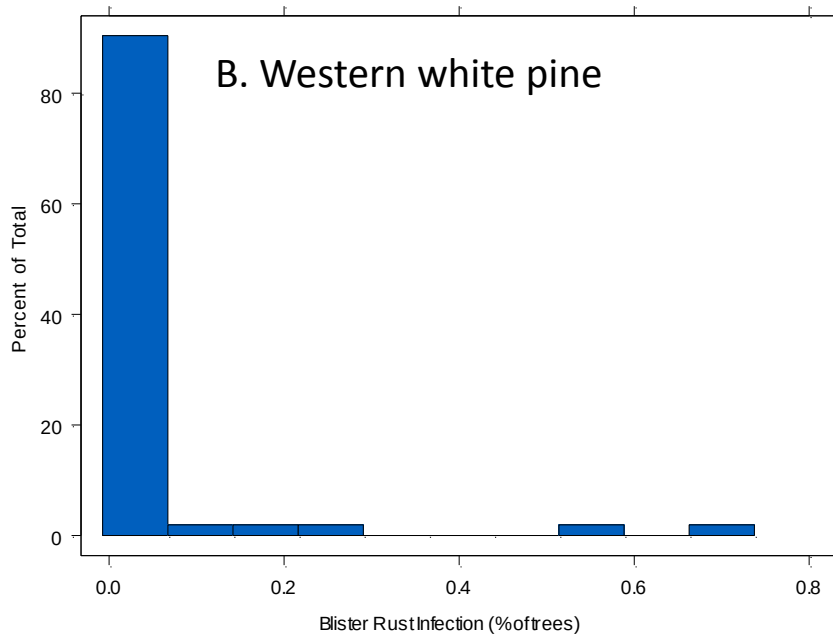
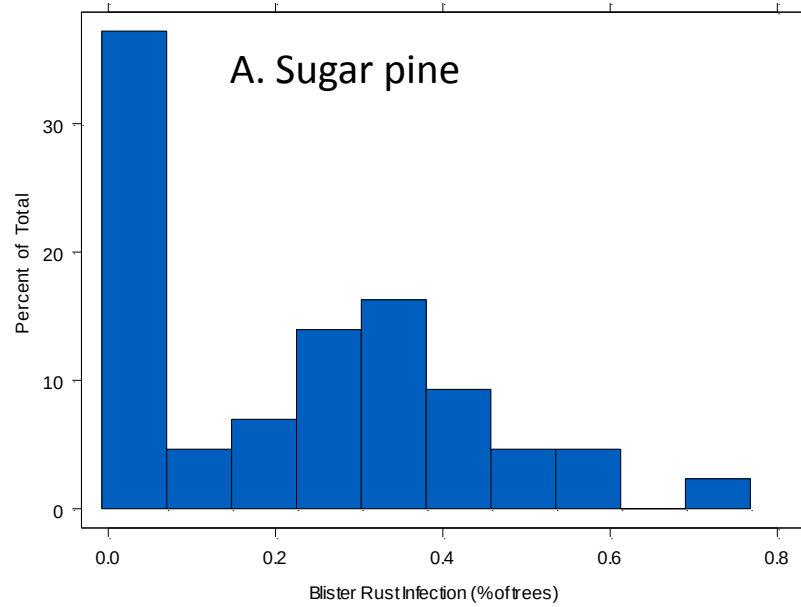


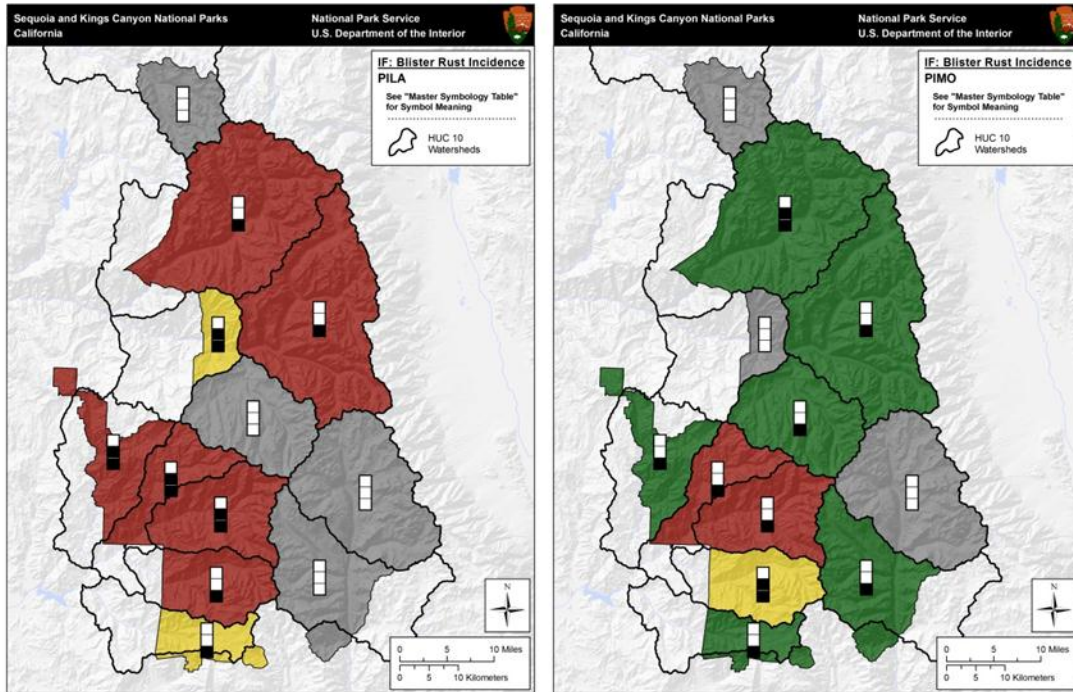
Figure 5. Distribution of blister rust infection rate for sugar pine (A) and western white pine (B) in Sequoia-Kings Canyon National Parks.

Table 9. Extent of white pine blister rust infection on sugar pine (PILA10) and western white pine (PIMO10) summarized by intact forest alliance at Sequoia-Kings Canyon National Parks. Values are the means (standard deviation) for all survey plots in each intact forest alliance. Data from Duriscoe and Duriscoe (2002).

Alliance	Incidence PILA10 (%)	Plots (#)	Incidence PIMO10 (%)	Plots (#)
Quaking aspen (2010)	--	--	--	--
Black oak (2020)	23.3 (11.1)	3	--	--
Lodgepole pine (3020)	--	--	0	2
Ponderosa pine (woodland, 3050)	--	--	--	--
Ponderosa pine (forest, 3060)	13.8 (8.3)	6	--	--
Giant sequoia (4020)	24.0 (10.8)	8	--	--
Mountain hemlock (4030)	--	--	--	--
Red fir (4050)	--	--	5.7 (4.7)	11
Red fir-White fir (4070)	43.7 (16.6)	3	70	1
White fir-Sugar pine (4080)	17.3 (2.4)	7	--	--
Incense cedar (4111)	--	--	--	--
Western white pine- Lodgepole (4540)	--	--	0	10

Table 10. Incidence of white pine blister rust in sugar pine (PILA10) and western white pine (PIMO10) in the intact forest at Sequoia-Kings Canyon National Park. Results pertain only to the intact forests in each HUC 10 watershed. Cell color codes reflect the criteria discussed in the text. Coding: green = “good”; yellow= “caution”; red = “concern.” The number of blister rust survey plots follows mean incidence in parentheses.

HUC 10	Incidence PILA10 (%)	Incidence PIMO10 (%)
Middle Fork Kings River	27.5 (2)	0 (7)
Upper South Fork San Joaquin River	no data	no data
East Fork Kaweah River	76.0 (1)	3.3 (3)
North Fork Kaweah River	20.8 (5)	0 (1)
Marble Fork Kaweah River	20.4 (5)	35.0 (2)
Lower South Fork Kings River	8.3 (3)	no data
South Fork Kaweah River	6.5 (2)	0 (1)
Golden Trout Creek-Kern River	no data	0 (2)
Upper South Fork Kings River	43.5 (2)	0 (1)
Rock Creek-Kern River	no data	no data
Middle Fork Kaweah River	26.0 (3)	25.0 (2)
Roaring River	no data	0 (2)



Map 11. Condition assessment of blister rust incidence on sugar pine (PILA) and western white pine (PIMO) for the intact forests in Sequoia Kings Canyon National Parks. Data source: Duriscoe and Duriscoe 2002. See Master Symbology Table in Appendix 3.

Assessment

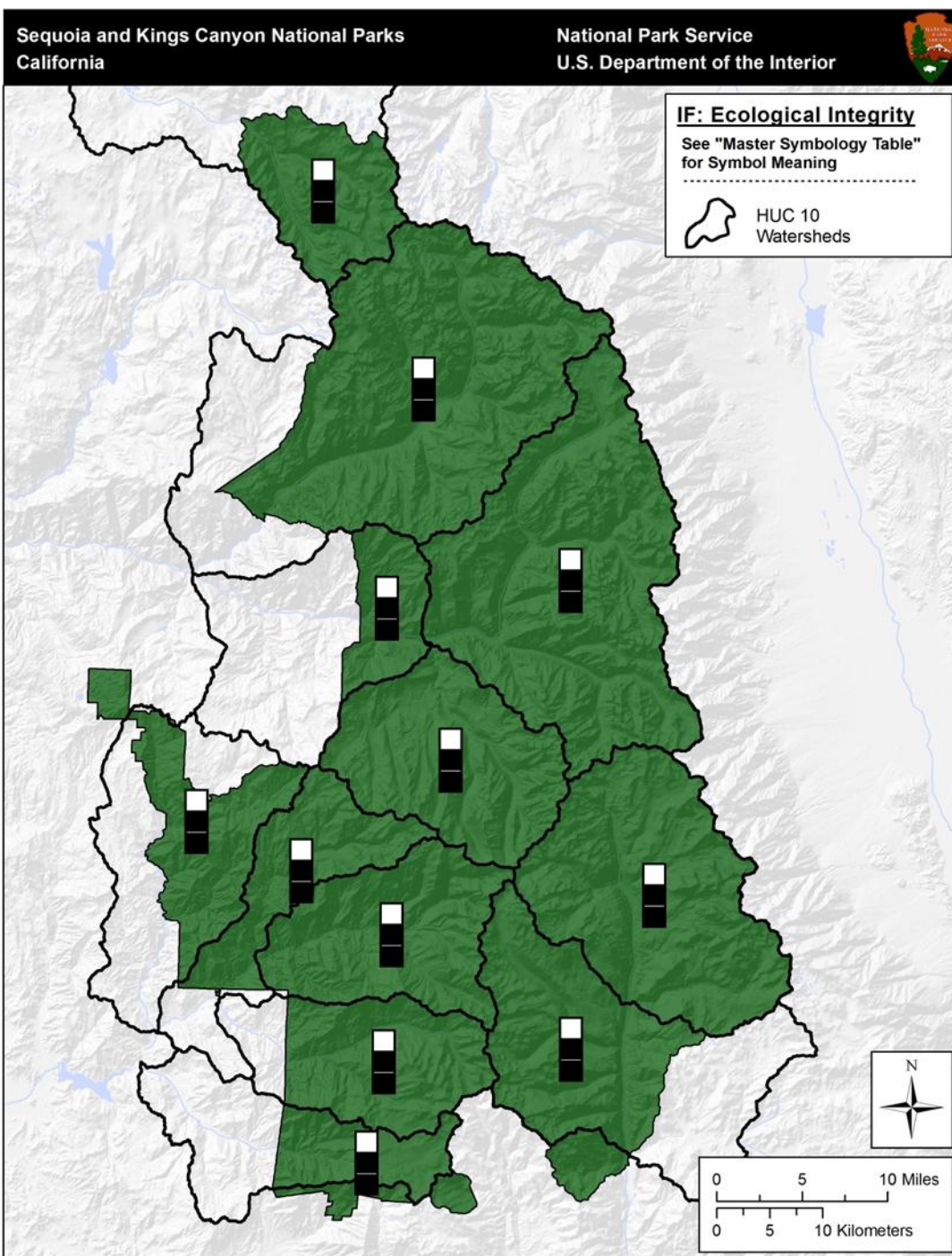
The intact forests had high ecological integrity throughout the Parks (Map 12). The forests were contiguous with large trees and high tree biomass. The size distributions suggested stable tree populations with a sufficient fraction of dead trees to provide critical wildlife habitat and input to the detrital carbon pool (Table 5). Comparisons with the regional forests indicated that SEKI forests are among the largest and most productive in the region (Table 2, Table 3).

However, our limited evaluation of trends suggested that the integrity of the intact forests may be deteriorating. Estimates of aboveground live tree biomass and big tree density declined for most forest alliances between the Stephenson inventory and NRI inventory (Table 6). Although we detected only one watershed (Map 6) suffering a drop in live tree biomass based on our statistical standard (non-overlap of means $\pm$ one standard deviation), the trend matches results from the demography plots. Between 1983 and 2004, tree mortality doubled while recruitment remained constant (van Mantgem and Stephenson 2007). Such demographic rates, if not offset by increased growth, imply lower biomass accumulation and smaller pools of live tree biomass.

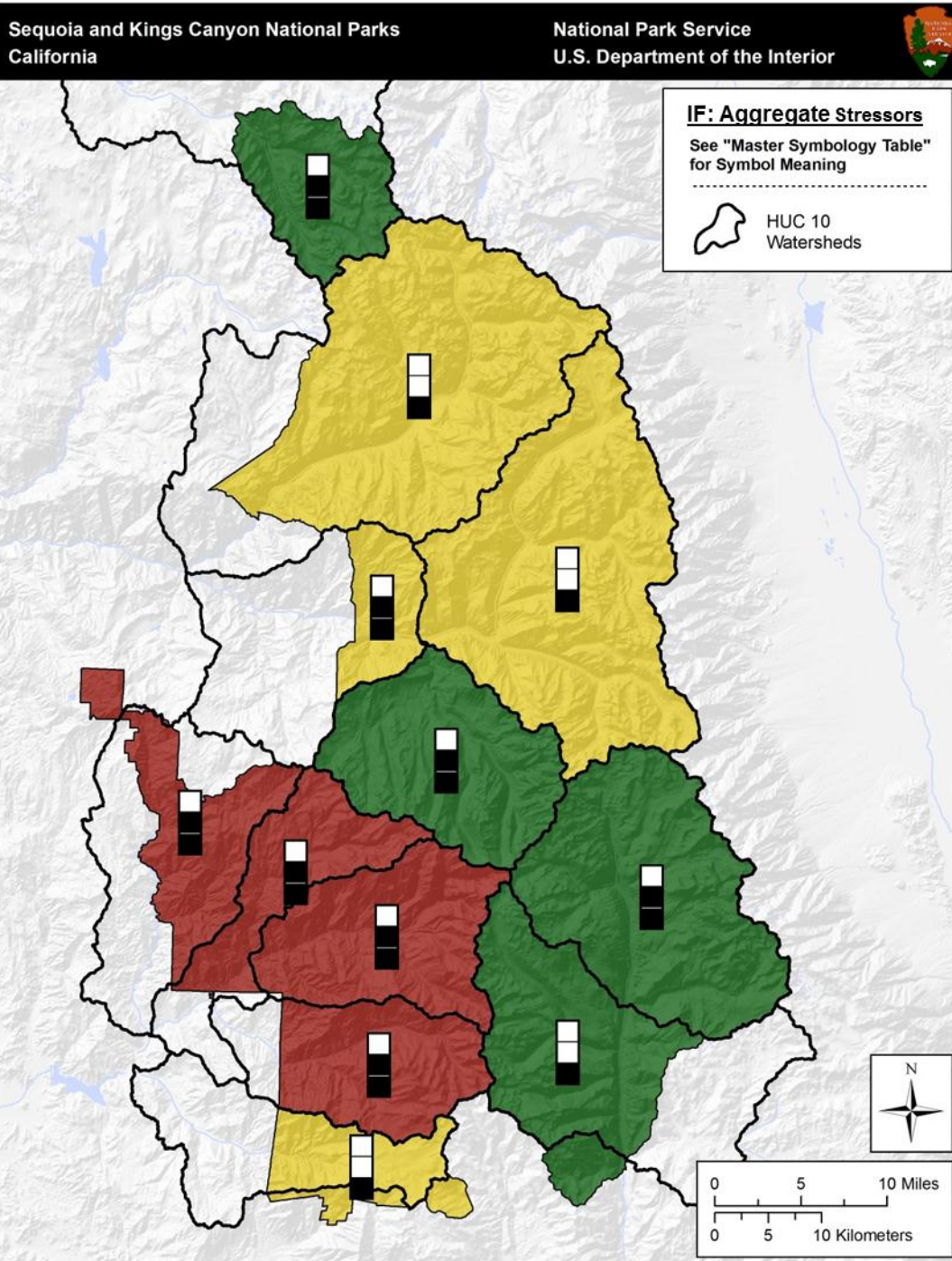
There is also a plethora of stressors that may impact the integrity of the intact forests. We averaged the extent of major stressors by HUC 10 watershed (Table 8, Table 10) to provide a simple aggregate view of the abiotic and biotic agents known to damage some component of the intact forest (Map 13). These agents (i.e., altered fire regime, air pollution, and blister rust) were concentrated in the Kaweah River drainage. The combination of an incipient decline in integrity and a battery of stressors is cause for concern.

Levels of Confidence In Assessment

Our estimate of ecological integrity was based on a scientifically robust, park-wide survey of vascular plants (NRI database, Graber and Haultain 1993). The plots were stratified to sample the diversity of physical gradients in the Parks. They met this criterion (Graber and Haultain 1993). However, they were not designed for spatial interpolation. In particular, plots were located in clusters in order to improve efficiency (Map 2). One result was that the more accessible intact forests in the Kaweah River drainage were oversampled, while the more remote forests in the Kings River drainage were undersampled. In the Kaweah, intact forests represented 49% by area, but accounted for 67% of the NRI plots. In the Kings River drainage, intact forests represented 41% of the area but only 22% of the NRI plots. The intact forests in the Kern were proportionately sampled. Our spatial interpolation attempted to mitigate this imbalance by summarizing results by forest alliance and by major drainage. However in absence of data, we used the park-wide average and this park-wide average included the oversampled Kaweah forest. Given these concerns we assigned a moderate level of confidence to our ecological integrity measures (Map 12).



Map 12. Natural resource condition assessment for intact forests in Sequoia Kings Canyon National Parks. Results based on the ecological integrity score developed in this report. See Master Symbology Table in Appendix 3.



Map 13. Vulnerability assessment for intact forests in Sequoia Kings Canyon National Parks. Results based on the aggregation of the major abiotic and biotic stressors developed in this report. See Master Symbology Table in Appendix 3.

The NRI sampling regime was more problematic in our assessment of trends. The NRI plots were measured over an 11-yr interval. Thus the temporal resolution was clouded. The combined spatial bias and temporal confusion made it difficult to definitively resolve trends. In addition, while Stephenson collected most of data in one year (1983), his plots were designed to examine vegetation-environment relationships. Thus Stephenson ensured that plots in rare environments were included (N. Stephenson, USGS WERC, personal communication). Such considerations add an unquantifiable amount of bias to average values. Consequently we assigned low confidence to our measure of trends in big tree density (Map 5) and aboveground live tree biomass (Map 6).

We used a variety of approaches to assign confidence to the major stressors. When available, we reported the confidence supplied by the information source (ozone exposure, nitrogen deposition). For blister rust, we applied a simple statistical rule of thumb that at least three samples are needed to estimate central tendency and variance. We considered any means with less than three samples to have low confidence. With three or more samples, we assigned moderate confidence. There was no quantitative basis to evaluate the confidence in the fire return interval departure (FRID) index. However it is a well-established and widely used metric. Consequently we attributed moderate confidence to mean FRID values.

Gaps in Understanding

Ecological integrity is a powerful concept that aspires to link resource monitoring results to resource management actions (Tierney et al. 2009). While the dimensions of integrity must be built on a sound scientific foundation, the application is informed by management goals and constrained by funding levels. The definition of integrity inherits some of the subjectivity and angst associated with defining the range of natural variability (Landres et al. 1999) or the reference condition (Stephenson 1999) for a particular population, ecosystem, or process. There are few if any absolute standards. For example, in our definition (Table 4), we considered large contiguous patches of intact forests as a positive contributor to integrity. This criterion follows a widely held and well supported relationship between the size of a patch and its value in supporting biodiversity (Lindenmayer et al. 2008). At the same time, large contiguous patches of forest cover increase the hazard associated with wildfires. In terms of fire behavior, larger patches represent a threat to integrity. How to balance these two perspectives (protect biodiversity; reduce fire risk) is not only a scientific question. Thus if ecological integrity is to become a useful standard, we need to develop a framework for integrating scientific insights and management priorities.

Another challenge is defining what should be measured to assess integrity. In the abstract, it is a relatively straightforward exercise. For example, net primary productivity (NPP) is an essential element of ecosystem function. NPP represents the amount of available carbon produced by the plants. It not only forms the base of the food chain but also fuels nutrient cycling. Along with climate, it is a required baseline measurement for all sites in the National Science Foundation's Long-Term Ecological Research network (<http://www.lternet.edu/>). At the species level, the key metric is population growth rate (λ). It forms the basis for population viability analyses and conservation planning (Wisdom et al. 2000). However these two variables require a major effort to measure. For parks the size of SEKI, direct measures of these parameters are impractical. The alternatives include indices that are feasible to measure. For example, we used live tree biomass and tree size distribution as proxies for NPP and population viability. While their value as indices was well justified, the relationships do not hold for all forests (Keeling and Phillips 2007, Wang et al. 2009). Therefore a key gap in our knowledge is the performance of these indices in Sierran forests. We also do not know how other alternatives to estimating key parameters of integrity might perform. For example in the northern hardwood forest of New England, field-parameterized models (Fahey et al. 2005) and remote sensing approaches (Smith et al. 2002) have proven to be viable options for estimating carbon and nitrogen flux at spatial scales relevant to managers.

We documented a host of factors that may negatively affect the integrity of some component(s) of the intact forests at SEKI. Initially we evaluated these stressors in isolation, and then, in absence of better information, we averaged their influence to get an idea of their potential aggregate impact on the forests (Map 13). However these stressors undoubtedly interact. Stress from air pollution can weaken trees and predispose them to damage by drought, insects, and disease, and the reverse can also occur (USEPA 1980, Allen and Brashears 1998, Miller and McBride 1999). At the physiological level, Panek (2004) established that for the western Sierra region, peak ambient ozone concentrations do not correspond with peak ozone uptake in ponderosa pine. Further, the deposition of nitrogen has the potential to impact nutrient uptake

and allocation in nitrogen-limited forests and modify tree response to stressors (Takemoto et al. 2001). The compounded effects of multiple stressors have caused dramatic shifts in community composition, trophic linkages, and overall ecosystem function in many communities (Paine et al. 1998) including forests dominated by a long-lived conifer (Eschtruth and Battles 2009). We need to know whether the stressors present at SEKI work in concert or opposition. We need to know if their impacts are additive or exponential.

Fire is a complex and crucial process in the intact forests. While the FRID is an excellent concept for reporting fire activity, it does not capture all the relevant information. Most importantly, FRID is a binary assessment of fire frequency – an area is burned or not. There is no measure of the severity of the fire. We must temper the tendency to assume all is well if the historical fire frequency is maintained (i.e., FRID values <2). A forest with a low FRID value as a result of an uncharacteristically high severity wildfire or as a result of an uncharacteristically low severity prescribed burn may be in a very unstable condition. Therefore we must also evaluate the ecological consequences of fire (Miller et al. 2009) to gain a complete understanding of the stress associated with an altered fire regime.

Recommendations for Future Research

Our recommendations for future research at SEKI are to remeasure the NRI plots and to integrate the available datasets. The value of remeasuring permanent plot data cannot be overstated. Despite our efforts to detect trends in key forest parameters, our results were compromised by spatial and temporal confusion. Among the available datasets, the NRI plots constitute the most extensive and least biased network. The baseline data is of very high quality and it is carefully archived. We found no inconsistencies in the dataset in the course of our analyses – a rare occurrence in our experience. Limitations in the spatial distribution of the plots (e.g., clustering to improve efficiency; oversampling of Kaweah forest) can be accommodated with statistical post-processing. The issue with timing can be addressed by securing the funds to remeasure all the plots in a much shorter interval.

The intact forests at SEKI are a very well-studied group of communities. There is a wealth of data available. The elements for an effective forest assessment program exist. However there seems to be limited efforts to make these data sets “talk to each other.” Granted, each data set was collected for specific a purpose with funding from a range of sources. Nevertheless there is unrealized potential to leverage past and current efforts.

The recently completed vegetation map provides the spatial coverage to interpolate plot-level results to the landscape. The long-term demography plots provide the temporal resolution to track vital rates for the major tree species that dominate the most common forest alliances. The NRI plots provide the extensive samples needed to accurately scale the rates measured in the demography plots to tree populations. The FIA plots, although sparse, provide spatially unbiased information that can be used to validate park-wide interpolations.

The challenge is to develop the quantitative structure that brings these disparate observations to bear on the question of ecological integrity. The goal is to achieve the best analysis, consistent with all the available information. We suggest that a fruitful approach to explore is formal data assimilation. Data assimilation is a family of techniques for improving estimates of geophysical and biophysical quantities combining models and observations. Although best known as a tool in weather forecasting and hydrological modeling, data assimilation is also used in the analysis of complex data sets. Recently the utility of these techniques have been demonstrated for quantifying the carbon dynamics of ponderosa pine forests in Oregon (Williams et al. 2005). The goal of a data assimilation program in regard to the intact forest should be to characterize the spatial variability of key forest metrics, detect incipient trends, and ultimately, increase the credibility of our assessments of ecological integrity.

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Appendix 1. Vegetation descriptions for the intact forest alliances excerpted from USGS-NPS (2007).

Quaking Aspen Forest Alliance (2010)

This forest alliance is mapped on gentle to very steep slopes of varying aspect between 1818 -- 3391 m. Stands of this type are restricted to areas of relatively high moisture availability, including steep talus slopes at the upper limits of its elevational range. The overstory is dominated by *Populus tremuloides* in either tree or tall shrub form, with *Abies concolor*, *Calocedrus decurrens*, *Pinus contorta*, or *Pinus jeffreyi* frequently contributing low cover. The shrub layer is often made up of *Artemisia tridentata*, *Chrysolepis sempervirens*, or *Salix* sp. The understory can be sparse or dense depending on moisture availability; more mesic sites often support a rich layer of mesophytic herbs. Areas dominated by the associations making up this alliance are usually subject to disturbance, such as avalanche or rock fall. Although substrate may be either metamorphic or igneous in origin, stands of *Populus tremuloides* appear to frequently occur on metamorphic parent material. Soils are moderately to well drained sands and sandy loams. The hydrology is upland.

California Black Oak Forest Alliance (2020)

This forest alliance is mapped on gentle to very steep slopes of varying aspect between 893 -- 2641 m. The overstory of these open to closed stands is dominated by *Quercus kelloggii*, with *Pinus ponderosa*, *Calocedrus decurrens*, *Pinus lambertiana* and/or *Quercus chrysolepis* often present. When present the shrub layer may include *Arctostaphylos patula*, *A. viscida*, *A. mewukka*, *Ceanothus* spp., or *Ribes roezlii*. Stands often have an understory of *Chamaebatia foliolosa* and a dense carpet of leaf litter, and are often found in late-seral post fire environments. Soils are moderately well drained sandy clay loams, loams, and sandy loams. The hydrology is upland.

Sierra Lodgepole Pine Forest Alliance (3020)

This forest alliance is mapped on a wide range of sites from gentle canyon bottoms to steep high slopes and ridges between 2028 -- 3523 m. Forests and woodlands included in this alliance are characterized by a closed to moderately open tree canopy that is dominated by *Pinus contorta* var. *murrayana*. *P. contorta* ssp. *murrayana* occupies a broad array of habitats in the Sierra Nevada, and this is reflected in the diverse range of associations it characterizes. Stands may be even- or multi-aged depending on geographic location, edaphic characteristics, and local fire history. Shrub and herbaceous layers may be present or absent depending on tree canopy characteristics and local site conditions. Stands are characterized by upland, palustrine, and seasonally flooded hydrology. Soils are well drained sands, loams, and sandy loams.

Ponderosa Pine Forest Alliance (3050)

This forest alliance is mapped on gentle to steep primarily south to southwest-facing slopes between 1165 -- 2125 m. The open tree canopy is often dominated by *Pinus ponderosa*, with *Abies concolor*, *P. lambertiana*, *Quercus chrysolepis*, and *Q. kelloggii* often occurring as co-dominants and *Calocedrus decurrens* sometimes contributing low cover. The shrub layer can be sparse or dense and may contain *Arctostaphylos patula*, *A. viscida*, *Ceanothus cordulatus*, *C. integerrimus* var. *californicus*, *Chamaebatia foliolosa*, and/or *Chrysolepis sempervirens* as

important species. The herbaceous layer is characteristically sparse. The hydrology is upland. Soils are well drained sandy loams.

Ponderosa Pine-Incense-Cedar Forest Alliance (3060)

This forest alliance is mapped on gentle to steep slopes of varying aspect (although primarily south to southwest or west facing) between 927 -- 2367 m. The open to closed tree canopy is dominated by a mixture of *Pinus ponderosa* and *Calocedrus decurrens*, with *Abies concolor*, *P. lambertiana*, *Quercus chrysolepis*, and/or *Q. kelloggii* frequently important. When mapped along water courses and on river terraces, *Calocedrus decurrens* may dominate the canopy, with *Alnus rhombifolia*, *A. concolor*, *Acer macrophyllum*, and/or *Torreya californica* also present. The shrub layer may be sparse or dense, with *Arctostaphylos mewukka*, *Arctostaphylos patula*, *Arctostaphylos viscida*, *Ceanothus cordulatus*, *C. integerrimus*, *Cornus sericea*, *Corylus cornuta* var. *californica*, *Chamaebatia foliolosa* and/or *Rubus parviflorus* contributing the highest amount of cover. The herbaceous layer is characteristically sparse, but may include dense patches of annual grasses or *Pteridium aquilinum*. The hydrology is upland or riverine. Soils are moderately well drained to well drained sandy loams or sandy clay loams.

Giant Sequoia Forest Alliance (4020)

This forest alliance is mapped on gentle to steep slopes of varying aspect between 1371 -- 2426 m on the western slope of the Great Western Divide. The tree canopy is dominated by a mixture of montane conifers, including *Abies concolor*, *Calocedrus decurrens*, and *Pinus lambertiana*, which provide a matrix within which *Sequoiadendron giganteum* provides an important component. A secondary layer of *Cornus nuttallii* is frequently present. The shrub layer is generally open and contains scattered patches of *Ceanothus integerrimus*, *Chrysolepis sempervirens*, *Corylus cornuta* var. *californica*, and young conifers. The herbaceous layer is generally sparse but can be locally well developed depending on microsite characteristics; it frequently includes *Adenocaulon bicolor*, *Draperia systyla*, *Galium* spp., *Lupinus polyphyllus* var. *burkei*, and *Pteridium aquilinum*. The hydrology is upland. Soils are moderately well drained sandy loams.

Mountain Hemlock Forest Alliance (4030)

This forest alliance is mapped on moderately steep to steep north to northeast facing slopes between 2535 -- 3302 m. The open, sometimes stunted tree canopy is dominated by *Tsuga mertensiana*, but may also have significant amounts of *Pinus contorta* var. *murrayana* and/or *P. monticola*. Low amounts of *Abies magnifica* or *P. albicaulis* may also be present. The shrub layer is well developed and contains a mesic understory of *Cassiope mertensiana*, *Ledum glandulosum*, *Phyllodoce breweri*, and *Ribes velutinum*. The herbaceous layer can be sparse or dense, and most frequently includes *Antennaria* spp., *Aquilegia formosa*, *Arabis platysperma*, *Aster breweri*, *Cryptogramma acrostichoides*, *Epilobium angustifolium* ssp. *circumvagum*, *Heuchera micrantha*, *Orthilia secunda*, and/or *Selaginella watsonii*. The hydrology is upland. Soils are well drained sandy loams to moderately well drained loams.

California Red Fir Forest Alliance (4050)

This forest alliance is mapped on gentle to steep slopes of varying aspect between 2003 -- 3275 m. The tree canopy of associations in this upper montane alliance is dominated by *Abies magnifica*. Other trees in the canopy may include *A. concolor*, *Pinus contorta* var. *murrayana*, *P.*

jeffreyi, *P. lambertiana*, and *P. monticola*. The shrub layer ranges from absent to well developed, and most frequently includes *Acer glabrum*, *Arctostaphylos nevadensis*, *A. patula*, *Artemisia tridentata*, *Ceanothus cordulatus*, *Chrysolepis sempervirens*, *Holodiscus discolor*, *Ledum glandulosum*, *Phyllodoce breweri*, *Prunus emarginata*, *Ribes spp.*, or *Symphoricarpos mollis*. The herbaceous layer is characteristically sparse to absent, with *Apocynum androsaemifolium*, *Arabis spp.*, *Aster breweri*, *Carex rossii*, *Elymus elymoides*, *Hieracium albiflorum*, *Lupinus latifolius* var. *columbianus*, *Monardella odoratissima*, *Pedicularis semibarbata*, *Pteridium aquilinum*, *Pyrola picta*, and *Senecio triangularis* among the most frequently encountered species. The hydrology is upland. Soils are moderately well drained to well drained sandy loams.

California Red Fir-White Fir Forest Alliance (4070)

This forest alliance is mapped on gentle to steep slopes of varying aspect between 1703 -- 3077 m. The tree canopy of associations in this montane alliance is dominated by a mixture of *Abies magnifica* and *A. concolor*. Other trees in the canopy may include *Juniperus occidentalis* var. *australis*, *Pinus contorta* var. *murrayana*, *P. jeffreyi*, *P. lambertiana*, and *P. monticola*. The shrub layer ranges from absent to well developed, and most frequently includes *Acer glabrum*, *Arctostaphylos patula*, *Ceanothus cordulatus*, *Chrysolepis sempervirens*, *Prunus emarginata*, *Ribes spp.*, or *Symphoricarpos spp.* The herbaceous layer is characteristically sparse to absent, with *Apocynum androsaemifolium*, *Arabis platysperma*, *Aster breweri*, *Elymus glaucus*, *Hieracium albiflorum*, *Lupinus spp.*, *Monardella odoratissima*, *Pteridium aquilinum*, and *Pyrola picta* among the most frequently encountered species. The hydrology is upland. Soils are well drained sandy loams.

White Fir-Sugar Pine Forest Alliance (4080)

This forest alliance is mapped on gentle to steep slopes of varying aspect between 1109 -- 2897 m. This forest alliance includes some of the most extensive montane forests mapped in the two parks, including stands of what is frequently referred to as the 'mixed conifer' forest. The tree canopy is dominated by a mixture of *Abies concolor* and *Pinus lambertiana*, with *Calocedrus decurrens* important on lower elevation sites. Other trees in the canopy frequently include *P. jeffreyi*, *P. ponderosa*, and/or *Quercus kelloggii*; *P. contorta* var. *murrayana* may also occur in colder air drainages. The shrub layer is dominated by regeneration of the coniferous overstory and may also contain *Arctostaphylos patula*, *Ceanothus cordulatus*, *Chamaebatia foliolosa*, *Chrysolepis sempervirens*, *Cornus nuttallii*, *Corylus cornuta* var. *californica*, *Prunus emarginata*, or *Ribes spp.* depending on site conditions. The herbaceous layer is characteristically sparse, but frequently includes *Adenocaulon bicolor*, *Apocynum androsaemifolium*, *Draperia systyla*, *Galium sparsiflorum*, *Hieracium albiflorum*, and/or *Pteridium aquilinum* in addition to young conifer seedlings. The hydrology is upland. Soils are well drained sandy loams.

Incense Cedar Forest Alliance (4111)

This forest alliance is mapped on gentle to moderately steep north facing lowslopes and canyon bottoms between 927 -- 2113 m. These stands are associated with watercourses and river terraces. The tree canopy is dominated by a mixture of *Calocedrus decurrens* and *Alnus rhombifolia*, with occasional contributions by *Abies concolor*, *Acer macrophyllum*, and/or *Torreya californica*. The sparse shrub layer is dominated by regenerating conifers and may also

contain *Cornus sericea*, *Corylus cornuta* var. *californica*, and *Rubus parviflorus*. The herbaceous layer is sparse to absent but may contain a mix of upland and wetland species. The hydrology is upland to riverine.

Western White Pine-Sierra Lodgepole Pine (4540)

This forest mapping unit is an aggregation of the *Pinus monticola* - *Pinus contorta* var. *murrayana* woodland association (3132) and the *Abies magnifica*-*Pinus monticola*-*Pinus contorta* var. *murrayana* woodland association (4055). The aggregated mapping unit is mapped on gentle to steep slopes of varying aspect between 2383 -- 3278 m. The open to moderately dense tree canopy is dominated by a mixture of *Pinus monticola* and *P. contorta* var. *murrayana*, with *Abies magnifica* important in lower elevation stands. *P. albicaulis*, and *P. balfouriana* ssp. *austrina* are also often present at low cover. The shrub layer is absent to well developed; when present, it is frequently dominated by *Arctostaphylos nevadensis* A. *patula*, *Chrysolepis sempervirens*, , various *Ribes* spp. and/or *Holodiscus microphyllus*. The herbaceous layer is sparse to absent, with *Arabis* spp., *Aster breweri*, *Elymus elymoides*, *Pteridium aquilinum*, *Carex exserta*, *C. rossii*, *Juncus parryi* and *Senecio triangularis* most common. The hydrology is upland. Soils are well drained sands or sandy loams.

Giant Sequoia Forest Alliance (4020)

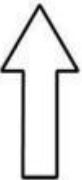
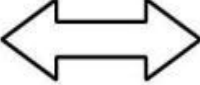
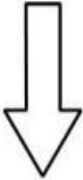
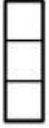
This forest alliance is mapped on gentle to steep slopes of varying aspect between 1371 -- 2426 m on the western slope of the Great Western Divide. The tree canopy is dominated by a mixture of montane conifers, including *Abies concolor*, *Calocedrus decurrens*, and *Pinus lambertiana*, which provide a matrix within which *Sequoiadendron giganteum* provides an important component. A secondary layer of *Cornus nuttallii* is frequently present. The shrub layer is generally open and contains scattered patches of *Ceanothus integerrimus*, *Chrysolepis sempervirens*, *Corylus cornuta* var. *californica*, and young conifers. The herbaceous layer is generally sparse but can be locally well developed depending on microsite characteristics; it frequently includes *Adenocaulon bicolor*, *Draperia systyla*, *Galium* spp., *Lupinus polyphyllus* var. *burkei*, and *Pteridium aquilinum*. The hydrology is upland. Soils are moderately well drained sandy loams.

Appendix 2

Species list for all trees found in Natural Resource Inventory database for Sequoia-Kings Canyon National Parks. The JCODE abbreviation and common name are referenced throughout this report.

JCODE	Genus	Species	Subspecies	Common name
ABCO10	<i>Abies</i>	<i>concolor</i>		white fir
ABMA10	<i>Abies</i>	<i>magnifica</i>	<i>magnifica</i>	California red fir
ABMA11	<i>Abies</i>	<i>magnifica</i>	<i>shastensis</i>	Shasta red fir
ACMA10	<i>Acer</i>	<i>macrophyllum</i>		big-leaf maple
AECA10	<i>Aesculus</i>	<i>californica</i>		California buckeye
ALRH10	<i>Alnus</i>	<i>rhombifolia</i>		white alder
CADE10	<i>Calocedrus</i>	<i>decurrens</i>		incense cedar
CEBE10	<i>Cercocarpus</i>	<i>betuloides</i>		birchleaf mountain mahogany
CEOC10	<i>Cercis</i>	<i>occidentalis</i>		western redbud
CONU10	<i>Cornus</i>	<i>nuttallii</i>		mountain dogwood
FRDI10	<i>Fraxinus</i>	<i>dipetala</i>		California ash
JUOC10	<i>Juniperus</i>	<i>occidentalis</i>	<i>australis</i>	Sierra juniper
PIAL10	<i>Pinus</i>	<i>albicaulis</i>		whitebark pine
PIBA11	<i>Pinus</i>	<i>balfouriana</i>	<i>austrina</i>	foxtail pine
PICO10	<i>Pinus</i>	<i>contorta</i>	<i>murrayana</i>	lodgepole pine
PIFL10	<i>Pinus</i>	<i>flexis</i>		limber pine
PIJE10	<i>Pinus</i>	<i>jeffreyi</i>		jeffrey pine
PILA10	<i>Pinus</i>	<i>lambertiana</i>		sugar pine
PIMO10	<i>Pinus</i>	<i>monophylla</i>		single leaf pinyon pine
PIMO20	<i>Pinus</i>	<i>monticola</i>		western white pine
PIPO10	<i>Pinus</i>	<i>ponderosa</i>		pacific ponderosa pine
PIPO11	<i>Pinus</i>	<i>ponderosa x jeffreyi</i>		
PLRA10	<i>Platanus</i>	<i>racemosa</i>		western sycamore
POBA11	<i>Populus</i>	<i>balsamifera</i>	<i>trichocarpa</i>	black cottonwood
POTR10	<i>Populus</i>	<i>tremuloides</i>		quaking aspen
PREM10	<i>Prunus</i>	<i>emarginata</i>		bitter cherry
QUCH10	<i>Quercus</i>	<i>chrysolepis</i>		canyon live oak
QUDO10	<i>Quercus</i>	<i>douglasii</i>		blue oak
QUKE10	<i>Quercus</i>	<i>kelloggii</i>		California black oak
QUWI10	<i>Quercus</i>	<i>wislizeni</i>	<i>wislizeni</i>	interior live oak
RHIL11	<i>Rhamnus</i>	<i>ilicifolia</i>		holly-leaf redberry
SASC21	<i>Salix</i>	<i>scouleriana</i>		Scouler's willow
SEGI10	<i>Sequoiadendron</i>	<i>giganteum</i>		giant sequoia
TOCA10	<i>Torreya</i>	<i>californica</i>		California nutmeg
TSME10	<i>Tsuga</i>	<i>mertensiana</i>		mountain hemlock
UMCA10	<i>Umbellularia</i>	<i>californica</i>		California bay

Appendix 3. Master symbology table used in condition assessment maps

STATUS				
	Good	Moderate	Poor	No Information
TREND				
	Improvement	No Change	Decline	No Information
CONFIDENCE				
	High	Medium	Low	No Information

Appendix 4

List of acronyms used in this report.

dbh	Diameter at breast height (1.37 or 1.4 m, depending on data set)
EPA	Environmental Protection Agency
FIA	Forest Inventory and Analysis
FRAP	Fire and Resource Assessment Program, California Department of Forestry and Fire Protection
HUC	Hydrological Unit Code
NAAQ	National Ambient Air Quality Standards
NPP	Net primary productivity
NPS	National Park Service
NRCA	Natural Resource Condition Assessment
NRI	Natural Resource Inventory
PACE	Protected-area centered ecosystem
ppb	Parts per billion
SEKI	Sequoia Kings Canyon National Parks
USGS	United States Geological Survey
WERC	Western Ecological Research Center
YOSE	Yosemite National Park

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

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