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Framework for monitoring shrubland community integrity in California Mediterranean type ecosystems: Information for policy makers and land managers

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Abstract

Shrublands in Mediterranean-type ecosystems worldwide support important ecosystem services including high levels of biodiversity and are threatened by multiple factors in heavily used landscapes. Use, conservation, and management of these landscapes involve diverse stakeholders, making decision processes complex. To be effective, management and land use decisions should be informed by current information on ecosystem quality and resilience. However, obtaining this information is often a challenge due to the extent of landscapes involved. Here we present a conceptual integrity monitoring framework based on simple easily observable ecosystem components readily understood by nonspecialists. Community integrity is defined by plant functional group based on relative proportion of shrubs and nonnative annual grasses. The ability to use these straightforward metrics results from four factors: relatively good alignment of characteristic bird, mammal, and insect communities with shrub cover, positive feedback between annual grasses and short fire intervals, the inhibitory effect of annual grasses on shrub seedling establishment, and similar functional group response to different disturbances. Two additional metrics, indicator species and shrub species diversity, capture subtle yet persistent signatures of disturbance on integrity not reflected in functional group composition. The framework is designed to: categorize habitats into ecosystem integrity classes, forecast likely integrity class changes caused by threats and environmental conditions, and provide a simple reporting mechanism that can be overlain with data on conservation status and vulnerabilities. The proposed framework includes a pilot phase to validate empirical relationships, thresholds, and sampling efficiency. The accessibility of these metrics to nonspecialists is anticipated to enhance communication among stakeholders and thus facilitate problem solving. Leveraging monitoring and mapping programs driven by other needs (e.g., species conservation and fire management) affords meaningful opportunities to offset program costs.

KEYWORDS

chaparral, coastal sage scrub, ecosystem integrity, monitoring, Mediterranean-type ecosystem shrublands

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1 | INTRODUCTION

Mediterranean-type ecosystems (MTEs) worldwide are recognized "hotspots" of biodiversity and support burgeoning human populations (Cincotta, Wisnewski, & Engelman, 2000; Cowling, Rundel, Lamont, Arroyo, & Arianoutsou, 1996). Shrubland communities comprise 34% of the land area of MTE in California and 52% of MTE's worldwide (Keeley, Bond, Bradstock, Pausas, & Rundel, 2012) and are thus an important conservation focus. In addition to biodiversity, shrublands provide multiple ecological services (Underwood, Safford, Molinari, & Keeley, 2018).

The ecological integrity of shrublands, that is the degree to which their structure, composition, and function operates within the bounds of historical variation, is threatened by a suite of interacting factors including altered fire regime (primarily increased fire frequency) (Keeley & Syphard, 2018), habitat loss and fragmentation (Underwood et al., 2018), invasive species (D'Antonio & Vitousek, 1992), land clearing (Stylinski & Allen, 1999), grazing (Hedrick, 1951), nitrogen deposition (Cox, Preston, Johnson, Minnich, & Allen, 2014; Pivovaroff, Santiago, Vourlitis, Grantz, & Allen, 2016), and climate change (e.g., extreme drought Jacobsen & Pratt, 2018; Park, Hooper, Flegal, & Jenerette, 2018). These threats compromise the ability of shrublands to provide important ecosystem services.

Further, wildland fire, a characteristic feature, threatens human communities. Growing human populations place high demands on these landscapes with only a fraction of the original habitat remaining (Underwood et al., 2018). Human developments convert and fragment remaining habitat leaving patches that may be too small to sustain target species and communities (Bolger et al., 1997; Lawson, Regan, Zedler, & Franklin, 2010). In addition, these patterns create wildfire risks by juxtaposing highly flammable vegetation and high-value properties while increasing fire ignition risk (Keeley & Syphard, 2018).

Use, conservation, and management of these landscapes include both high ecological complexity and high social complexity due to past conflict among stakeholders over diverse uses (e.g., resource use, biodiversity conservation, fire management) (Gill & Stephens, 2009). Creating effective communities of practice (Amin & Roberts, 2008) is essential to support policy and management solutions. Clear, accessible information on community integrity and resilience is needed to ensure knowledge exchange among stakeholders, build trust, support conflict resolution, and balance conservation and sustainable landscape use (Lawson, Hall, Yung, & Enquist, 2017). However, there is not a generally accepted measure of shrubland integrity, vulnerability, and resilience.

We present an integrity monitoring framework based on ecosystem components readily understood by nonspecialists, expected to enhance communication among policy makers, resource users, land managers, and scientists. In our framework, community integrity is defined by the relative proportions of plant functional group composition (Figure 1). In addition, select indicator species capture subtle yet persistent signatures of disturbance not reflected in functional group composition (Lucas, Johns, Jiang, & Yang, 2013). The ability to use these simple metrics results from (a) relatively good alignment of characteristic bird, mammal, and insect communities with shrub cover, (b) positive feedback between annual grasses and short fire intervals, (c) the inhibitory effect of annual grasses on shrub seedling establishment, and (d) similar response of shrub and annual grass cover to the primary disturbances in this ecosystem (Diffendorfer et al., 2007). We considered use of all annual invasives but did not for clarity and simplicity. Annual grasses are easily distinguished from a distance and have important system impacts, including fostering short fire intervals, not shared by other common invasives (e.g., *Brassica* spp, *Hirschfeldia* spp, and *Erodium* spp). The framework includes: categorizing habitats into integrity classes, forecasting likely integrity class changes caused by threats and environmental conditions, and providing a simple reporting mechanism (annual maps) that can be overlain with data on conservation status (e.g., endangered species status) and vulnerabilities (e.g., short fire intervals).

2 | METHODS

We used information from the literature on ecosystem function and dynamics to identify indicators and preliminary thresholds that characterize shrubland integrity as the foundation for our framework.

2.1 | MTE shrubland ecology

MTE shrubland communities' array along an aridity gradient with drought deciduous dominated coastal sage scrub (CSS) on drier sites (e.g., south aspects), and evergreen dominated chaparral on wetter sites (Poole & Miller, 1975). The communities also transition along a disturbance gradient where fire-disturbed chaparral can shift to a greater cover of CSS species (Keeley & Keeley, 1988). Coastal stands of both CSS and chaparral support higher native floristic diversity (Axelrod, 1978; Vasey, Loik, & Parker, 2012). CSS can intergrade with chaparral and can increase in canopy gaps in old chaparral stands (Hanes, 1971; Keeley & Keeley, 1988). In the absence of disturbance, the native closed canopy shrublands in southern California are relatively resistant to invasion by exotic species, in part due to their dense cover.
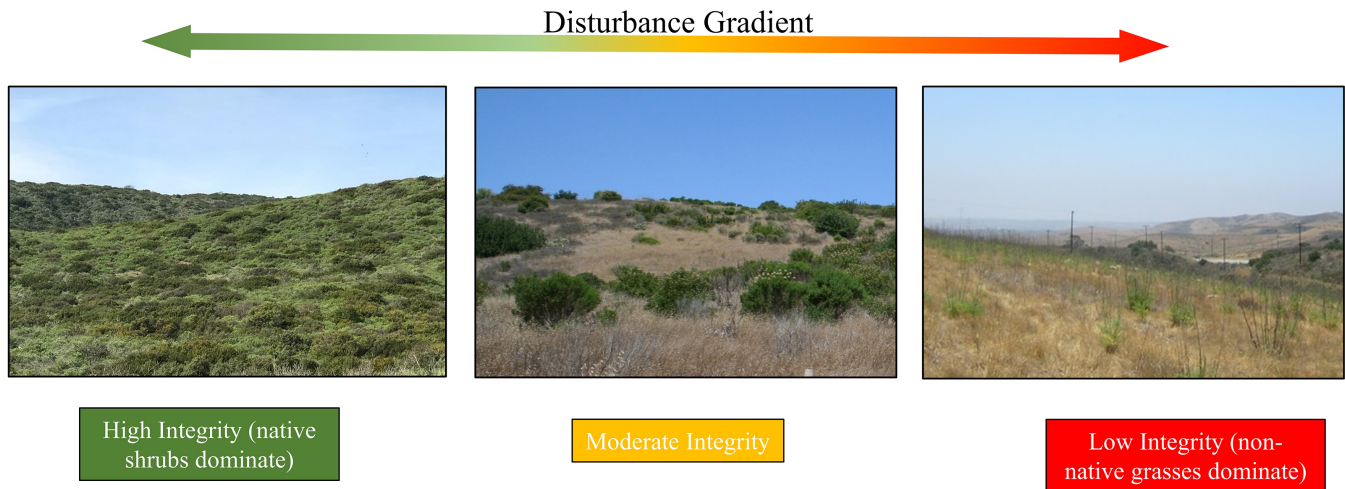
Disturbance Gradient

High Integrity (native shrubs dominate)

Moderate Integrity

Low Integrity (non-native grasses dominate)

FIGURE 1 Integrity classes are defined by plant functional group composition (proportion of shrub and annual grass). Visual distinction between these functional groups makes them readily understood by nonspecialists. Photo credits Dawn M. Lawson

These communities are resilient to periodic wildfire, though they are more accurately described as adapted to a particular fire regime, with fire return interval the most important factor (Keeley & Safford, 2016). This is a critically important feature of this ecosystem.

Much is known about the disturbance, degradation, and recovery dynamics in CSS and chaparral (Diffendorfer et al., 2002). In short, the naturally fire-prone vegetation is resilient to infrequent fires through persistent seed and bud banks. Highly drought-resistant postfire resprouts and seedlings promote rapid vegetation recovery after periodic wildfire. The main driver of stand degradation has been overly frequent fire where communities reburn before seed and bud banks replenish (Keeley & Brennan, 2012). Invasion by annual grasses contributes to a grass-fire cycle wherein short fire intervals foster annual grass invasion which fosters short fire intervals (D'Antonio & Vitousek, 1992). Drought can also restructure communities. Because drought operates at a larger spatial scale than fire, extreme drought, even in the absence of fire, could potentially shift community species composition to favor deeper-rooted species (Venturas et al., 2016) with large effects across the landscape. Then, because postfire patterns of species establishment are strongly influenced by prefire patterns of individual plants within a stand, these shifts could be perpetuated by fire (Odion & Davis, 2000). Livestock grazing impacts have been implicated in shrubland degradation but are poorly studied (Diffendorfer et al., 2002) and difficult to succinctly characterize because they vary widely based on the species of livestock (due to forage preference and behavior), and frequency, intensity, duration, and season of grazing (Rinella & Hileman, 2009). This is further complicated by the influence of site characteristics (e.g., aridity) on shrubland resilience (Diffendorfer et al., 2002).

Large-scale changes are often referred to as vegetation type conversion (Syphard, Brennan, & Keeley, 2018). The degree of permanence and reversibility of type conversion has important policy and management implications. On a practical level, falsely assuming that thresholds exist, beyond which changes are difficult-to-reverse, may lead to unnecessary interventions. Alternatively, the existence of unidentified thresholds may mean missed opportunities to prevent ecosystem damage (Suding & Hobbs, 2009). For CSS and chaparral, the reversibility of state changes (e.g., vegetation type conversion) and reversal rate appears to vary over their geographic range. There is evidence in the literature of both reversible (over several decades) state changes in more coastal sites (DeSimone & Zedler, 1999; Gressard, 2012), and possibly irreversible state changes in more inland sites (Minnich & Dezzani, 1998; Stylinski & Allen, 1999). Degraded stands that do not recover over time do not necessarily reflect the presence of thresholds if repeated disturbance maintains the degraded state (Gressard, 2012) and recovery is not delayed once the disturbance stops.

2.2 | What is ecosystem integrity?

Ecosystem integrity has been defined as the degree to which the structure, composition, and function of an ecosystem reflect historical variation (Karr & Chu, 1999). Disturbance outside the range of historical variation drives changes in integrity. High integrity means an ecosystem approaches natural structure, composition, and function; low represents substantial degradation relative to high integrity.

A practical way to define this gradient for MTE shrublands uses the proportion of vegetative cover composed of exotic annual grasses (Diffendorfer et al., 2007).

Although there are other invasives in these systems, exotic grasses have the most important impact on integrity because of their influence on:

1. Structure: They promote short fire intervals (D'Antonio & Vitousek, 1992) and compete with shrub seedlings during establishment (Eliason & Allen, 1997), driving declines in woody shrub cover.
2. Composition: They alter composition of plants and shrub associate taxonomic groups (Kluse & Doak, 1999).
3. Function: They drive increases in fire frequency through increased fuel loading, fuel continuity, and probability of ignition when dry (Keeley, 2002).

Vulnerability refers to the ease and likelihood of a stressor resulting in a decline in ecosystem integrity. Sites of high to intermediate integrity are vulnerable to degradation from repeat fire until seed and bud banks have recovered (Keeley et al., 2012). Resilience, or the ability to rebound to a preexisting condition after change, is a positive quality at the high end of the integrity spectrum, but negative (resisting an increase in integrity) or neutral (resist further degradation) at the low end (Briske, Fuhlendorf, & Smeins, 2005). In the context of shrubland monitoring and management this term should be used only with respect to specific sites and stressors (e.g., physical disturbance, drought, altered fire regimes, invasives, and climate change). Over the long term, climate change is anticipated to slowly degrade MTE shrubland resilience (Jacobsen & Pratt, 2018; Park et al., 2018).

2.3 | Integrity classification system

2.3.1 | Indicators of ecosystem integrity

Disturbance outside the range of historical variation drives changes in integrity and integrity classification should be based on species or functional group response to disturbance (White & Pickett, 1985). A range of taxonomic groups was considered for potential inclusion in this framework, but the scope was narrowed to vegetation because it correlates reasonably well with other taxa including characteristic bird, mammal, and insect communities that for the most part, align with shrub cover (Diffendorfer et al., 2007). Vegetation has a well-documented relationship to disturbance (Diffendorfer et al., 2002). Annual grass cover both (a) increases as a result of disturbance and promotes further disturbance (e.g., wildfire) and (b) impedes shrubland recovery. Native shrub cover correspondingly declines as disturbance increases. The ability to use vegetative characters to represent multiple taxa results in a simplified classification system, easier to implement, and easier to understand.

For this framework, absolute cover of woody vegetation, invasive annual grass cover, and shrub density were chosen.

Even though young shrub stands have very different cover characteristics than mature stands, due to autosuccessional processes shrub seedling and resprout density in young stands can be used to project shrub cover in mature stands (Hanes, 1971). The proposed preliminary density thresholds (Data S1) in recently burned stands are based on field studies that link stand age and seedling density (Cario & Zedler, 1995).

2.3.2 | Thresholds in shrubland integrity

A classification scheme to be useful, must reduce complexity and clarify meaningful patterns. In systems where they occur, ecological thresholds represent points where community responses to threats, natural ecosystem drivers, and/or management, change (Suding & Hobbs, 2009). States on either side of these thresholds vary in composition and structure and typically persist when the original disturbance is removed or abated. For those systems that exhibit them, thresholds and the states they define are a useful framework for ecosystem integrity. Disturbance is required to push the system across a threshold but once a community shifts, the alternative state may persist indefinitely unless perturbed again. Stability is a defining characteristic of states separated by ecological thresholds but this does not mean that the states are fixed and the degree of stability and resilience can vary (Beisner, Haydon, & Cuddington, 2003).

Resilience, the ability to recover to previous conditions after disturbance, often declines as thresholds are approached. For example, following a long free fire interval a chaparral stand has sufficient seed and resprouts to restock the canopy, but if burned, it's resilience to a subsequent fire is low until the seedbank has recovered. When ecological thresholds are crossed the relationships between drivers and ecological properties change from linear to nonlinear so that a small change in a driver results in a much larger response than at other places along the response curve (Briske et al., 2005).

In addition, thresholds are characterized by whether ecosystem changes are reversible with either natural or anthropogenic perturbations (Sasaki, Furukawa, Iwasaki, Seto, & Mori, 2015; Suding & Hobbs, 2009). Some irreversible thresholds are crossed when nonrenewable resources such as soils are lost, changing the ecosystem's capacity to support species previously present. Invasive species can create difficult-to-reverse thresholds by changing competitive relationships. When competitive relationships are altered, changes are not irreversible if environmental stochasticity creates periodic conditions where previous species assemblages are competitive and can regain space. Although important for ecosystem management, the reversibility of thresholds is generally poorly defined for most systems (Suding & Hobbs, 2009).

Reversible thresholds are often characterized by a time lag (hysteresis) after an ecosystem driver reverts to historical values during which the system is slow to recover (Beisner et al., 2003; Suding & Hobbs, 2009). Thus, threshold drivers may need to be defined separately for degradation and recovery processes. For example, a much longer fire interval may be required for stand recovery than would be required for maintenance of a high integrity stand. This is because once shrubs and their propagules are gone from a patch, propagules must disperse in and several generations of seed production, and recruitment may be required to fill gaps in shrub cover.

While the theory is well developed, identifying thresholds in practice remains elusive because thresholds often involve nonlinear changes in multiple parameters, and multiple interacting natural and anthropogenic drivers (Groffman et al., 2006). Extensive datasets including experimental evidence can be required to establish them conclusively (Beisner et al., 2003). While definitively establishing their existence may be problematic, the development of threshold (or state and transition) models based on heuristics has proven useful (Bestelmeyer, Brown, Trujillo, & Havstad, 2004; Briske et al., 2005). Suding and Hobbs (2009) recommend evaluating existing data for abrupt transitions, sharp spatial boundaries, interactions among drivers, and feedbacks that control recovery and resilience to determine if the system appears to exhibit threshold dynamics. If it does, they recommend developing a preliminary model and validating it as it is used.

Evidence of thresholds and alternate stable states in MTE shrublands comes from studies of short fire intervals and grazing, and recovery from cultivation. In the mid-1900s a focus on livestock production led scientists to experiment using fire to "type convert" shrublands to grasslands to increase forage for livestock (Hedrick, 1951; Sampson, 1944). This early work revealed that while shrub cover was resilient to relatively long fire-free intervals, short intervals could be used to rapidly reduce or eliminate shrub cover and that while the changes were not irreversible, recovery was delayed in these stands. Annual grasses were sometimes seeded into young stands to increase fine fuels and support more complete burns to achieve higher shrub seedling mortality (Burcham, 1955). More recently, Zedler, Gautier, and McMaster (1983) documented a sudden state change with sharp spatial boundaries from shrub dominated chaparral and CSS to an annual grass and forb dominated community after a short fire interval.

Other evidence of thresholds is associated with farming and land clearing. Even after cultivation ceases shrublands recover slowly (Tierra Data Systems [TDS], 2002) or not at all (Stylinski & Allen, 1999). Close to the coast these sites can achieve 50–90% shrub cover (TDS, 2002) but limited diversity. Farther from the coast fallowed sites may not regain more than 10% shrub cover of native shrubs decades after disturbance stops (Stylinski & Allen, 1999). While these sites can have high conservation value (e.g., support the threatened California gnatcatcher), they do not recover shrub diversity.

We developed state and transition models for CSS and chaparral where the states are equivalent to integrity classes in this framework and fire interval is the key driver that forces shifts across key shrubland thresholds (Data S1). Preliminary threshold values (Data S1) were developed from the literature and expert opinion. Other drivers in addition to fire and invasive annual grasses that are anticipated to affect threshold dynamics include weather patterns influencing moisture availability (Williams, Hobbs, & Hamburg, 1987), site position with respect to the aridity gradient (Poole & Miller, 1975), anthropogenic nitrogen deposition (Cox et al., 2014), and grazing (Diffendorfer et al., 2002).

2.3.3 | Shrub species diversity

While shrub cover is a good metric in most cases, certain disturbances, particularly farming, can result in stands with persistently low diversity even after recovery of shrub cover (TDS, 2002). Thus, formerly cultivated sites with only *Artemisia californica* and *Baccharis pilularis* will be classified as intermediate if total shrub cover is greater than 41%, or low integrity if cover is less than 41%.

Although a native shrub, the presence of *Malosma laurina* in very high densities may represent another form of type conversion in CSS. As a strong resprouter, it is favored by short fire intervals that can eliminate shrubs that regenerate from seed and extreme drought which disproportionally harm shallower rooting shrubs (Venturas et al., 2016). Stands of 80% or more *M. laurina* will be classified as intermediate integrity.

3 | RESULTS

3.1 | Monitoring framework

The framework is designed to: (a) categorize habitats into ecosystem integrity classes, (b) forecast integrity vulnerabilities due to threats and environmental conditions (e.g., drought), and (c) provide a simple reporting mechanism (annual maps) that can be overlain with data on conservation status (e.g., endangered species presence) and vulnerabilities (e.g., short fire interval). It includes baseline integrity mapping, integrity mapping updates, and vegetation monitoring to create and validate the integrity maps and updates, and to validate and refine the integrity classification system and at-risk overlays (Figure 2). It integrates data at
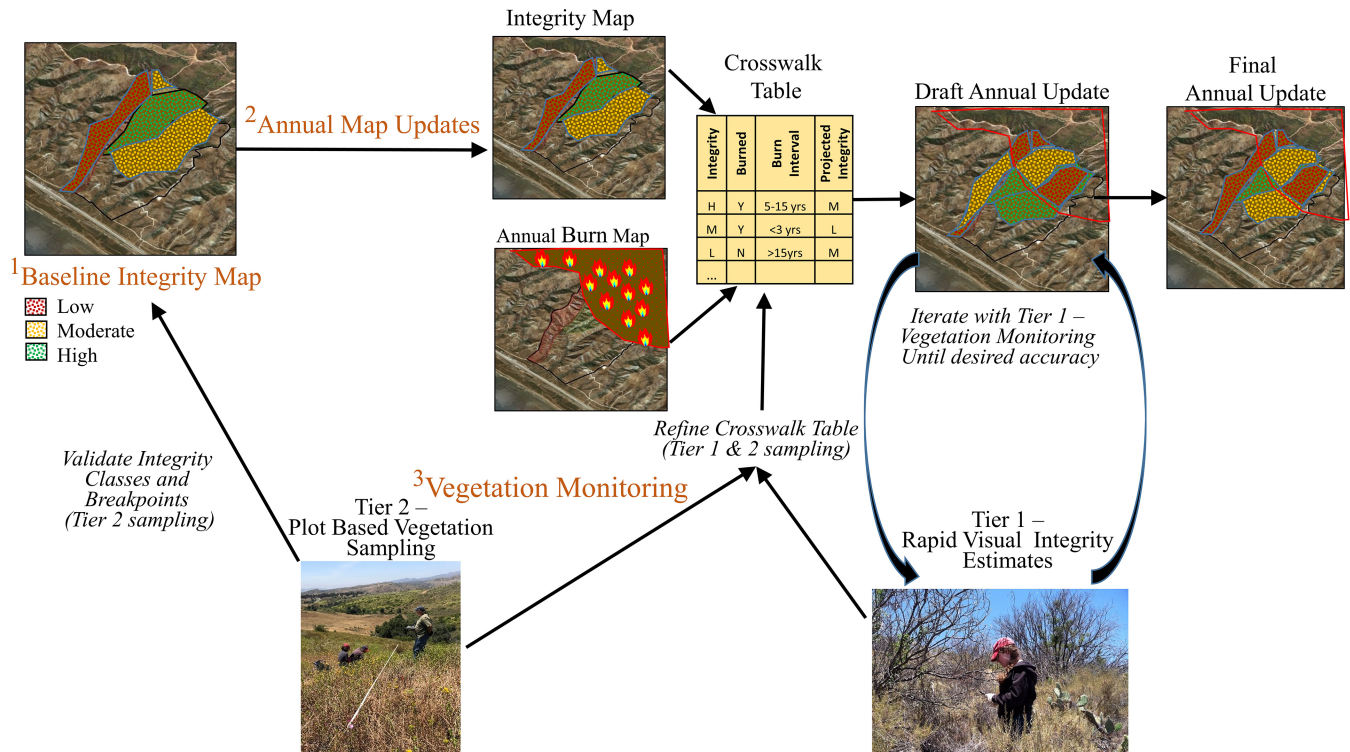
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FIGURE 2 The monitoring framework includes baseline integrity mapping, map updates, and vegetation monitoring. The process starts with the baseline integrity map, the annual or periodic updates use spatial fire data, and a cross-walk table capturing what is known about the effect of fire on integrity to produce a draft update. (Ultimately information on other drivers will also be included.) The draft is validated and refined using Tier 1 rapid assessments. Tier 2 sampling is used both to refine the cross-walk table and to validate integrity classes and breakpoints. Photo credits Dawn M. Lawson

three spatial scales: (a) landscape, (b) habitat patch, and (c) transect. Map and integrity classification system validation and refinement utilizes a two-tiered vegetation sampling system with Tier 1 employing rapid visual estimation techniques and Tier 2 using plot-based measurements. A pilot phase is incorporated to adjust sample size to ensure adequate but not excessive field efforts. Regular review and revision are designed to improve the monitoring system, addressing key uncertainties as knowledge accrues.

3.1.1 | Integrity mapping

Integrity mapping is the foundation of the framework. The baseline integrity map can be generated using various methods including remote sensing (Park et al., 2018) and field-based vegetation mapping. To use vegetation mapping the classification scheme must include percentage cover of shrubs and invasive grasses. Annual updates of integrity maps utilize annual wildland fire spatial data to project integrity changes using hypothesized integrity class transitions based on fire history (Data S1). While fire is the primary disturbance in these habitats, over time as the contributions of other disturbances and environmental factors to transitions are better understood, the integrity class

transition model (Data S1) will be refined to include those factors.

3.1.2 | Vulnerability overlays

Vulnerability overlays identify which sites are at risk of integrity decline based on year of last fire and the hypothesized transitions shown in Data S1. Vulnerability overlays may also be envisioned and developed for other threats based on stakeholder needs.

3.1.3 | Conservation value overlays

Overlays identifying particular conservation values are intended to be developed as needed to assist decision-makers. These are anticipated to differ among land management units based on organizational values and management drivers (e.g., at-risk species, stands with exceptionally long fire-free intervals).

3.1.4 | Vegetation sampling

To identify errors, improve map accuracy, and improve projections of integrity changes, a two-tiered vegetation

sampling system is proposed. Tier 1 utilizes rapid visual estimation techniques to validate the baseline map, create annual updates, and reevaluate transition rules (Data S1) used for map updates. Tier 2, plot-based monitoring, will validate the integrity classification system, including threshold values, and be used to improve predictions of integrity changes based on disturbance and environmental factors that influence time to recovery (Cox et al., 2014; Deutschman & Strahm, 2011; Diffendorfer et al., 2007; Keeley, Fotheringham, & Baer-Keeley, 2006; Vasey et al., 2012). Additionally, experiments may be employed when it is determined to be more cost-effective than observational monitoring. The goal is not long-term monitoring per se, although long-term monitoring may take place to answer specific questions.

3.1.5 | Prioritization of stands for management

Both landscape scale inventories of integrity and improved understanding of threshold dynamics are anticipated to support management prioritization. As knowledge of threshold dynamics accrues, the ability to identify management needed to prevent irreversible or difficult-to-reverse ecosystem damage should increase. However, prioritization will also reflect individual land owner values and mandates.

3.1.6 | Data management

The data management system must address quality assurance and efficiently produce a complete, accurate database to support timely analysis. A well organized and documented publicly available database is needed to support collaboration among related programs and facilitate unanticipated uses of the data, maximizing the cost-effectiveness of the program.

3.1.7 | Pilot phase

In the pilot phase of protocol implementation, both Tier-1 and Tier-2 vegetation sampling will be done across gradients (e.g., precipitation, fog, aspect, fire) and is anticipated to take 2–5 years. These data will be used to clarify empirical relationships, validate threshold presence and preliminary values (Data S1), and improve the efficiency of map updates. Variance components analysis and power analyses will be conducted with this data to ensure sampling intensity is high enough to support timely, reliable integrity assessments while not wasting resources oversampling (Deutschman & Strahm, 2011).

3.2 | Links to other monitoring and mapping programs

In addition to the obvious connection with vegetation mapping, efficiencies may be gained by linking this work with other regional monitoring and mapping programs. Fuel type mapping is one such program. Fuel types are vegetation assemblages defined by structure and predicted fire behavior (Scott, Burgan, & Robert, 2005). In MTE shrublands, fuel types are defined specifically by shrub and grass composition (Technosylva, Inc., 2014). These data, with some modification of collection methods, translate directly to the ecosystem integrity metrics proposed and could be used to generate baseline maps and map updates. A second example is the California Gnatcatcher South Coast Regional Monitoring Program. Data collected by this program could potentially substitute for or augment both Tier-1 and Tier-2 vegetation monitoring in CSS communities (https://sdmmp.com/view_project.php?sdid=SDID_201612021615.5).

4 | DISCUSSION

MTE shrublands exist in some of the most intensively used landscapes on the planet. Large areas have been converted to human land uses, primarily urbanization, and agriculture. The remaining lands provide important ecosystem services in their natural state and are important for biodiversity conservation, and recreation. However, these natural landscapes can burn in conflagrations, threatening human communities and ecosystem services. Efforts to understand, mitigate, and manage fire risk are a high priority. To further complicate matters, current threats (e.g., short fire intervals, habitat loss, and fragmentation) are anticipated to interact with future changes in climate (Battler, Parisien, Krawchuk, & Moritz, 2013; Park et al., 2018) in ways that are difficult to anticipate. Complex land ownership and land use patterns bring diverse stakeholders into land use decisions. Given the number of trade-offs and range of stakeholders involved in the use and conservation of MTE shrublands in California, easily accessible, timely data on ecosystem integrity is needed to support strategic management that balances land and resources use with ecosystem sustainability.

The existence of simple metrics (shrub and annual grass cover) that capture complex ecological dynamics and represent ecosystem integrity and are easy for nonspecialists to discern in the field is both remarkable and fortuitous. Monitoring these metrics, to be sustainable, needs to be easy to repeat, cost effective, and timely. We propose a monitoring framework that includes three spatial scales, a pilot phase, monitoring, and databases and archives. The two-tiered sampling program allows simplified rapid monitoring to support landscape scale data and more detailed patch and plot scale

data to better characterize apparent ecological thresholds, degradation, and recovery dynamics and refine the classification system. This knowledge as it is developed will support improved models of degradation and recovery and thus streamline updates of landscape scale integrity maps. In addition, plot-based sampling can be used to address questions not specifically related to refining the classification system. The pilot phase includes validation. While this framework focuses on vegetation, studies could be initiated for other taxa to better characterize their response to the disturbance gradient and vegetation metrics used here. Then the condition of those taxa could be inferred based on monitoring results under this framework.

Resources to implement this program represent a formidable challenge. Building communities of practice (Lawson et al., 2017) will likely be necessary to effectively execute this monitoring protocol. Shrublands exhibit high variability in both composition and function over both space and time (Axelrod, 1978; Deutschman & Strahm, 2011). In addition, with threats, ecosystem functional processes, and ecosystem services acting at a landscape scale no one land holder may have the ability to fully address questions regarding thresholds and recovery and degradation dynamics. Collaboration among landowners will likely be necessary to reduce costs and develop a full understanding of ecosystem dynamics within a region. Public availability of the data will improve the ability to address management and ecological questions faster with more certainty and reduce the cost to individual land owners.

Additionally, leveraging other regional mapping and monitoring programs provides an opportunity to economize. Integrating with these efforts may be difficult because the programs represent significant efforts to bring multiple stakeholders together to develop goals and objectives, and address trade-offs needed to optimize programs. Revisiting decisions may involve resistance as additional resources (time and money) will be needed and the people involved also have other priorities. But existing monitoring and mapping programs that generate very similar data (e.g., fuel type mapping) should be leveraged to develop this important information. This monitoring framework represents a significant opportunity to develop decision support knowledge that is widely accessible to stakeholders in these valuable and sought-after ecosystems. However, significant investment both in human capital and resources will be needed to achieve the desired goals.

CONFLICT OF INTEREST

The authors declare no potential conflict of interest.

AUTHORS CONTRIBUTIONS

D.M.L. and J.E.K conceived and designed the study, were responsible for drafting and revising the manuscript, conducted analysis and interpretation of the data, provided final approval of version to be published, and are accountable for all aspects of the work.

DATA AVAILABILITY STATEMENT

Data will be published separately with USGS.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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