Quantifying biological integrity of California sage scrub Communities using plant life-form cover

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Quantifying Biological Integrity of California Sage Scrub Communities Using Plant Life-form Cover

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Abstract

The California sage scrub (CSS) community type in California's Mediterranean-type ecosystems supports a large number of rare, threatened, and endangered species, and is critically degraded and endangered. Monitoring ecological variables that provide information about community integrity is vital to conserving these biologically diverse communities. Fractional cover of true shrub, subshrub, herbaceous vegetation, and bare ground should fill information gaps between generalized vegetation type maps and detailed field-based plot measurements of species composition and provide an effective means for quantifying CSS community integrity. Remote sensing is the only tool available for estimating spatially comprehensive fractional cover over large extent, and fractional cover of plant life-form types is one of the measures of vegetation state that is most amenable to remote sensing. The use of remote sensing does not eliminate the need for either field surveying or vegetation type mapping; rather it will likely require a combination of approaches to reliably estimate life-form cover and to provide comprehensive information for communities. According to our review and synthesis, life-form fractional cover has strong potential for providing ecologically meaningful intermediate-scale information, which is unattainable from vegetation type maps and species-level field measurements. Thus, we strongly recommend incorporating fractional cover of true shrub, subshrub, herb, and bare ground in CSS community monitoring methods. Estimating life-form cover at a 25 m x 25 m spatial scale using remote sensing would be an appropriate approach for initial implementation. Investigation of remote sensing techniques and an appropriate spatial scale; collaboration of resource managers, biologists, and remote sensing specialists, and refinement of protocols are essential for integrating life-form fractional cover mapping into strategies for sustainable long-term CSS community management.

Introduction

California's Mediterranean-type ecosystems, which cover 320,000 km^2 (Rundel, 1998), contain over 5,000 native plant species including at least 1,500 endemics (Remjanek et al., 1998). The sage scrub plant community type of California and Baja California (known as California or coastal sage scrub; CSS; Figure 1) occurs in a semi-arid portion of the ecosystem, which experiences moderate winter rainfall (250-450 mm) and warm to hot, dry summer (26-37ºC). The community type is dominated typically by aromatic, low-statured, facultatively deciduous, non-sclerophyllous (i.e. soft-stemmed) shrub species (known as subshrub), mixed with evergreen sclerophyllous (i.e. hard-stemmed) shrubs (also known as true shrub), and herbaceous plants (Axelrod, 1978). CSS communities provide habitat to at least 375 rare, threatened, or endangered species (Westman, 1981). Thus, the CSS community type has critical values for conservation (Rundel, 2007).

The CSS community type is one of the most disturbed ecosystems in the southern California region (Hierl et al., 2008; Noss et al., 1995; Westman, 1981). CSS communities are increasingly fragmented and endangered because of human-induced and altered disturbance including urban development, increase fire frequency, exotic plant invasion, and intense grazing and recreational activities (Keeley and Keeley, 1984; Minnich and Dezzani, 1998; Talluto and Suding, 2008; Westman, 1981). Presently less than 15% of the original habitat remains (Davis et al., 1994; O'Leary, 1990; Westman, 1981). To protect these communities from further loss and degradation, monitoring change in internal conditions of vegetation communities (known as biological/ecological integrity) is vital to understanding status and trend. Effective and efficient monitoring protocols (e.g. selecting appropriate variables and sampling methods) that incorporate comprehensive quantification of ecological integrity are urgent needs for sustainable long-term community management (Nichols and Williams, 2006).

Currently two approaches representing endpoints of spatial and biological/categorical scales are utilized for monitoring CSS communities: vegetation type mapping and field sampling of individual plants (Table 1). Vegetation community type maps represent spatial distributions of plant community type in a spatially comprehensive manner. A vegetation type map is a powerful tool for providing information about areal extent, distribution, and spatial arrangement of community types (Franklin and Woodcock, 1997). However, it is not sufficient for detecting habitat degradation when monitoring conditions within a community. Plot-level data recording species occurrence, abundance and richness can provide detailed information about internal conditions of a community. Because CSS communities are very heterogeneous (Rundel, 2007), obtaining reliable information based on plot-level data would require a large number of sample plots, which are expensive and time-consuming to measure. Therefore, there is a need for monitoring variables which fill the information gap between vegetation type maps and field-based data by characterizing vertical and horizontal structures within a community.

The primary goal of this paper is to inform the effectiveness of fractional cover at the plant life-form level (i.e. true shrub, subshrub, herb, and exposed bare ground) for quantifying conditions within CSS com-

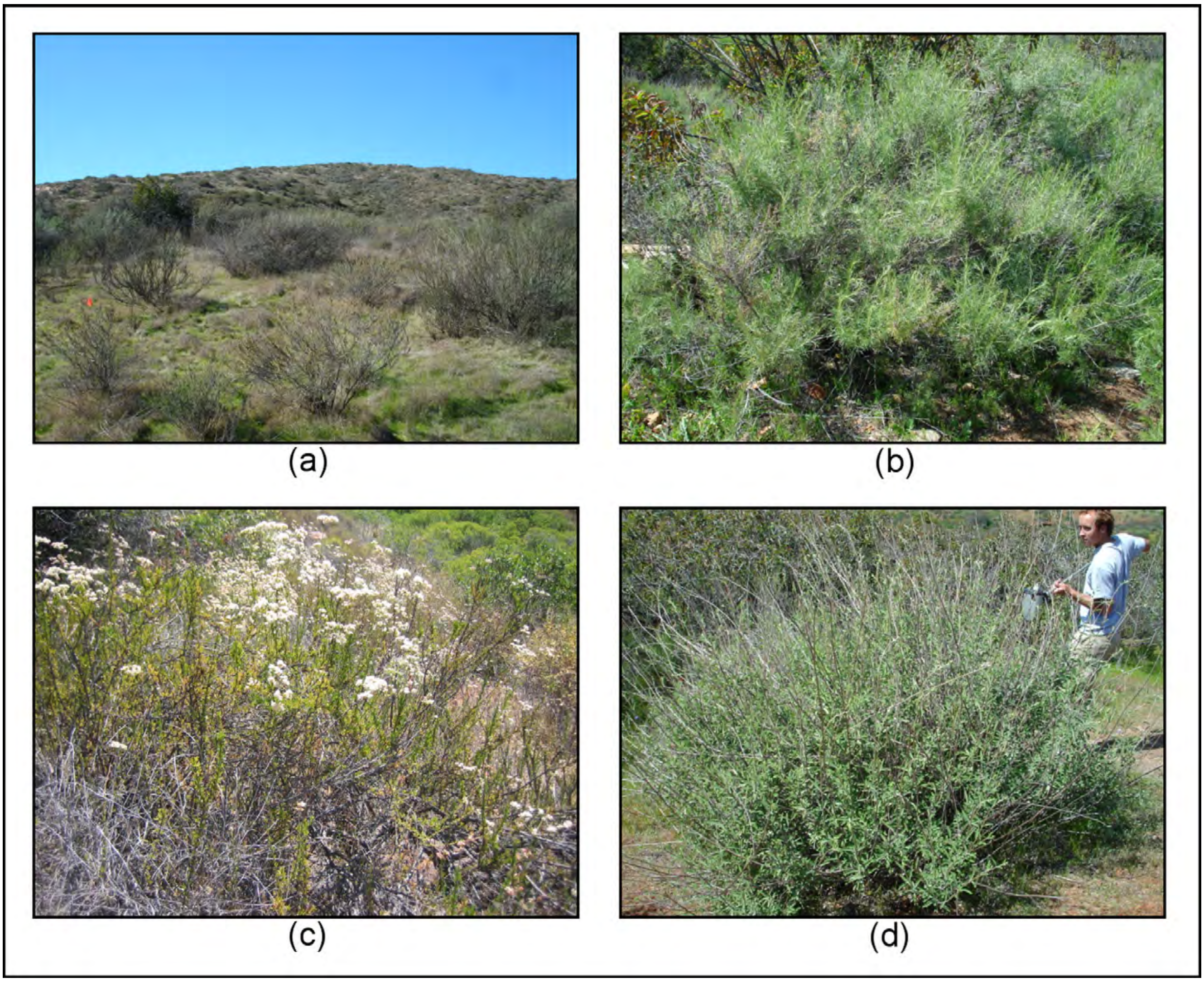

Figure 1 California sage scrub community (a) landscape and three common subshrub species: (b) *Artemisia californica* (California sagebrush), (b) *Eriogonum fasciculatum* (flat-topped buckwheat), and (c) *Salvia mellifera* (black sage).

Table 1 Current CSS community monitoring methods

Method	Community Type Mapping	Field-based Surveys
Biological Scale	Community type	Species
Spatial Scale	Coarse (e.g. > 0.5 km^2)	Fine (e.g. 1 m^2)
Sampling Density	High (i.e. Wall-to-wall)	Low (i.e. Sparsely distributed sampling plots)
Cost	Expensive	Moderately expensive
Temporal Frequency	Decadal	Annual

munities. To accomplish this goal, we synthesize information extracted from scientific studies, experimental investigations, and ecological conservation programs in CSS communities of southern California, as well as arid and semi-arid shrublands and rangelands that are physiognomically similar to the CSS community type. Objectives include: (1) examine the utility of life-form cover as a monitoring variable for arid and semi-arid vegetation communities, (2) review approaches for estimating life-form cover, and (3) investigate appropriate spatial scales for estimating life-form cover. Concepts developed in this paper can also be used to develop or refine monitoring strategies for shrubland communities in other Mediterranean-type ecosystems including: (1) shrubland in the Karoo, South Africa where increased fire frequency and biodiversity loss from grass invasion are primary concerns (Rahlao et al., 2009), (2) semi-arid savanna in Australia where expansion of woody life-forms in native grassland is a critical issue (Fensham et al., 2005), and (3) semi-arid Mediterranean shrubland in the Mertola region of South Portugal where landscapes have gone through grassland-shrubland transformation after agro-pastoral practice ceased (Castro and Freitas, 2009).

Cover for Quantifying Ecosystem Integrity

A common dilemma in vegetation community monitoring is limited financial and personnel resources (Godinez-Alvarez et al., 2009). The fundamental trade-off between cost and benefit is inevitable, and variable selection is "among the most difficult and controversial" steps in developing a monitoring program (Noss et al., 1997). Along with species composition, cover is the most frequently utilized monitoring variable for many terrestrial ecosystems (Godinez-Alvarez et al., 2009).

What is Cover and Why Cover?

Mueller-Dombois and Ellenberg (1974, p. 80) define cover as "the vertical projection of the crown or shoot area of a species to the ground surface expressed as a fraction or percent of a reference." The terms projected foliage cover (PFC) (Coops and Culvenor, 2000; Graetz, 1990; Munro et al., 2009), foliage projected/projective cover (FPC) (Hunter, 2005; Lucas et al., 2006), and projected canopy cover (PCC) (Fiala et al., 2006; Holmes, 1990; Williams and Hunt, 2002) are used in literature. PFC, FPC and PCC are defined as the proportion of the cover of foliage or canopies that are projected to the surface per unit area (Stow, 1995). The difference between canopy and foliage cover is substantially greater for woody life-forms than herbaceous life-forms. Figure 2 illustrates potential difference in shrub cover between PFC and PCC. Because of the small leaf size of shrub species in CSS communities, a shrub canopy more likely contains measurable gaps between foliage. Measurements of cover based on PFC could be considerably less than those based on PCC (Graetz, 1990). Therefore, estimates of shrub cover could substantially vary depending on the definition of 'cover.'

Cover is recognized to have greater ecological significance than density, which is obtained by total number of individuals divided by a unit area, because cover is correlated to biomass more strongly than the number of individuals (Mueller-Dombois and Ellenberg, 1974). Cover is an unbiased quantitative measure regardless of plant size (Bauer, 1943) or life-forms (Mueller-Dombois and Ellenberg, 1974). Cover at the life-form level can also serve as a proxy of vertical structure of a community (Graetz, 1990). Cover is an appropriate variable for monitoring CSS communities, many shrublands, and grasslands experiencing shrub invasion because those communities consist of multiple life-forms.

Cover of Plant Life-forms

Fractional cover of vegetation life-forms (also known as growth form) refers to general classes for plant species or individuals that are grouped based on their similarities in structure, function, and growth dynamics (Mueller-Dombois and Ellenberg, 1974). Plant life-form or functional group is an effective biological level for quantifying cover for the following reasons. The concept of ecosystem integrity is often defined in terms of ecosystem function. Because multiple species may perform comparable functions (King, 1993; O'Neill et al., 1986; Vitousek et al., 1997), differences

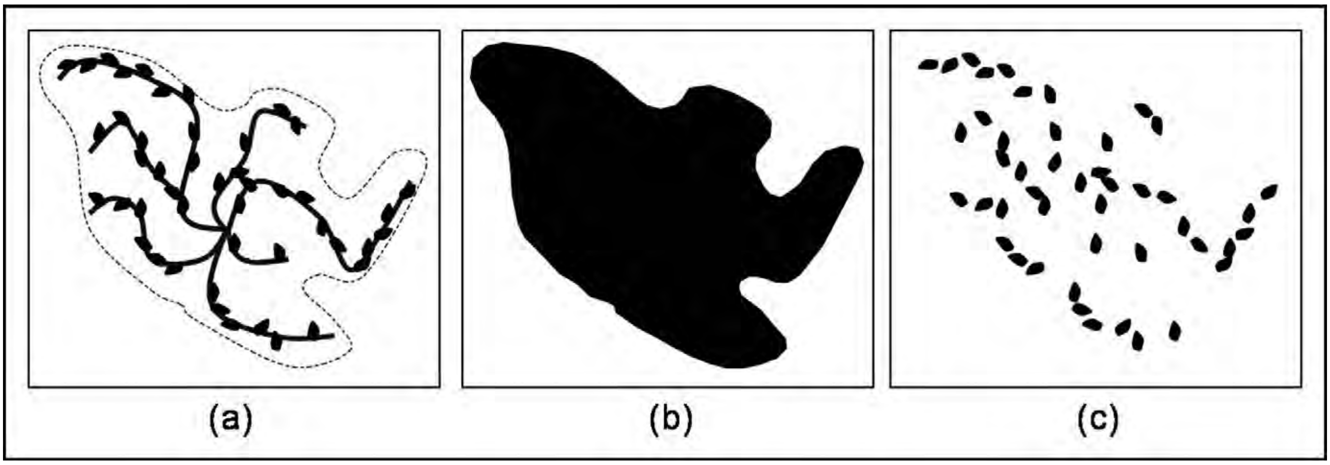

Figure 2 Illustration of projected canopy cover (PCC) and projected foliage cover (PFC) of a hypothetical subshrub. (a) Schematic drawing of a subshrub canopy. The canopy extent is delineated in a dash line. (b) PCC perspective. (c) PFC perspective. Target (canopy or foliage) cover is shown in black. Modified from Witztum and Stow (2004).

in species-based cover may have very little impact on whole system function. Variation or fluctuation at a species level is usually attenuated at a community/ ecosystem level (King, 1993). Mueller-Dombois and Ellenberg (1974) noted life-form as a primary element for structure; thus, life-form is likely to be a meaningful biological level for quantifying community integrity from a structural perspective. In fact, Westman and O'Leary (1986) indicate that true shrub and subshrub cover are effective indicators of the recovery of overall vegetation structure in CSS communities following disturbance. Grouping species based on their life-form types is a common practice in many ecological studies pertaining to its function (Graetz, 1990). Since plant life form is a finer biological scale than vegetation community type, a combination of life-form cover and vegetation type maps should provide information about what conditions of CSS community patches are distributed and how those patches are configured in the region.

Cover can be measured more reliably and cost-effectively at the life-form level than the species level. Because species are distributed along environmental gradients individualistically or independently (Gleason, 1939), heterogeneous and floristically diverse CSS communities would require a very large number of samples to estimate cover at the species level. Thus, estimating cover at the species level is not feasible for spatially comprehensive long-term ecological and rangeland monitoring.

From the perspective of an ecological hierarchy, variables at higher levels such as life form are often considered to be more stable than those at lower levels such as species (Carignan and Villard, 2002). Incorporating life-form level monitoring along with species surveys and vegetation type mapping into monitoring protocols also meets the requirement for multiple scale measures for quantifying ecological integrity that have been recommended by White and Bratton (1980), Woodley and Theberg (1992), and Andreasen et al.(2001).

Approaches for Estimating Life-form Cover

Two general approaches—field sampling and remote sensing—can be used to estimate fractional cover of life-forms. Table 2 shows common field sampling methods for *in situ* estimates of foliage cover based on direct measurement or visual estimation. Because of the unique characteristics of each method, cover estimates can vary (Bauer, 1943; Brakenhielm and Qinghong, 1994; Godinez-Alvarez et al., 2009). Carlsson et al. (2005) and Smith (1944) also reported that cover estimates in shrub- and grass-dominant communities can substantially vary across observers. Deutschman (2008) reported that variation in cover across sampling plots within a site contributes to approximately 25 to 30% of total cover variation at a life-form level when analyzing a total of 28 samples based on four sampling sites, each of which containing three sample plots over multiple visits. Thus, a large number of plots are required for reliable estimates for a community.

Needs for frequent monitoring imposes another challenge on field sampling particularly for spatially heterogeneous temporally dynamic ecosystems, which are subject to disturbances such as fire. Because of floristic dynamics of many semi-arid vegetation communities, frequent surveying is necessary, especially the first few years following fire in order to monitor habitat recovery. Close monitoring of herbaceous cover is also critical to assessing habitat degradation associated with exotic plant infestation (D'Antonio and Vitousek, 1992; Keeley et al., 2005; O'Leary, 1990). In order to estimate fractional cover using field methods, frequent access to permanent plots is inevitable. Thus, the vegetation within the plots and surrounding areas would potentially degrade. Frequent sampling of a large number of plots

Table 2 Common field sampling methods to estimate fractional cover

Method	Measurement basis	Reference
Visual estimate	Apparent area occupied by each cover type within a unit area.	O'Leary & Westman 1988, Herrick et al. 2006, Deutschman et al. 2008
Point intercept	The number of "hits" for each cover type at points along linear transect lines.	Herrick et al. 2006, Deutschman et al. 2008
Line intercept	The length/distance covered by each cover type along linear transect lines.	Westman 1981, Malanson 1984, Herrick et al. 2006
Quadrat	Dominant cover type within a quadrat or within each plot placed in the quadrat.	Keeley & Keeley 1984, Westman & O'Leary 1986, Deutschman et al. 2008

is more likely infeasible even though direct observation in the field is known to provide accurate and precise cover estimates.

Remote sensing provides an effective means for estimating fractional cover while compensating for the drawbacks of field-based methods (Vogiatzakis et al., 2006). Remote sensing is the collection of data without physical contact with phenomena or objects of interest, by quantifying electromagnetic radiation reflected or emitted from the earth's surface (Greegor, 1986; Jensen, 2007). Remote sensing is known as an indispensable tool for ecological studies over large areas (Graetz, 1990; Kerr and Ostrovsky, 2003; King, 1993). Fractional cover of plant life-form types is one of the measures of vegetation state that is most amenable to remote sensing (Graetz, 1990; Shoshany, 2006). While remote sensing estimates of vegetation cover are normally derived from airborne or satellite imagery, close-range remote sensing techniques such as hand-held or very-low altitude spectral radiometry provides a means for making detailed, low spatial density estimates (Jensen, 2007). Figure 3 shows a very high spatial resolution image of a part of CSS communities. Different cover types appear different in color, tone, and/or texture. This canopy perspective (or top-down vertical viewing angle) of remote sensing yields more reliable estimates of cover, which is measured as vertically projected area than would a oblique or quasi-horizontal viewing angle (Witztum and Stow, 2004). Optical remote sensing is particularly suitable for the Mediterranean-type ecosystems because of the number of cloud-free days in the regions (Stow, 1995).

Wall-to-wall coverage of large areal extents by remote sensing systems enables comprehensive cover estimates with unbiased sampling and a larger sample size by surveying entire communities. Historic archives of remotely sensed data permit retrospective assessment of communities and thus, are suitable for long-term monitoring (Washington-Allen et al., 2006). Semi-automated data processing for remotely sensed imagery can provide a cost-effective means of reliably estimating fractional cover over large areas and more frequently, which is desirable for monitoring highly fragmented, spatially heterogeneous, and temporally dynamic arid and semi-arid communities.

Limitations of cover estimates using remote sensing are that such measurements are indirect, and the spatial sampling unit (i.e. pixel) is arbitrary. In remote sensing, spectral reflectance signals from elements on the ground are assumed to be isolated from environmental and instrumental noise (Stow, 1995). Further, targets are assumed to be spectrally separable from background, and different target types are assumed to have unique spectral signatures (Friedl et al., 2001). Highly reflective bare ground in arid and semi-arid regions that obscure signals from vegetation is a well-known obstacle to accurately quantifying vegetation cover (Okin et al., 2001; Shoshany, 2000; Witztum and Stow, 2004). During wet seasons, both shrubs and herbaceous plants tend to have high reflectance in the near infrared (NIR) spectral region, while herb and bare ground have comparably high reflectance throughout the visible and NIR spectral range during dry seasons. Thus, the spectral separability between shrub and herb, or herb and bare ground can be particularly problematic in arid and semi-arid environments.

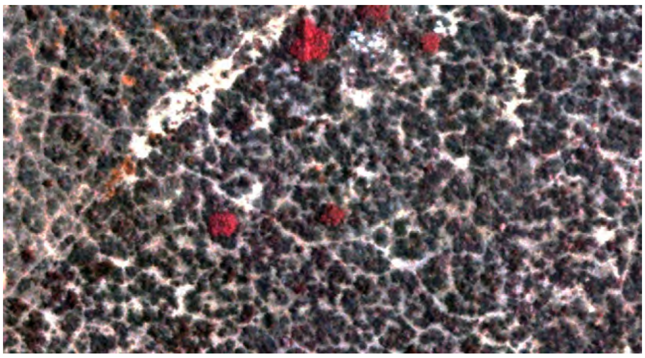

Figure 3 Subset of scanned color-infrared aerial photography of a California sage scrub community type acquired on July 22, 2005. The spatial resolution of the image is 0.15 cm, and near infrared, red, and green spectral bands are displayed in red, green, and blue color guns, respectively. True shrubs appear as large red circular objects at the center and on the top middle part of the image, and subshrub patches appear dark greenish-gray clumps throughout the scene. Herbaceous plants and bare ground occur between shrub patches and appear in orange tone in light gray tone, respectively.

Pixels are arbitrarily imposed spatial sampling units that not only obscure boundaries between elements in a scene but likely contain more than one ground element. Mixtures of cover types within a pixel impact delineation of objects of interest such as shrub canopies and herbaceous patches, which would increase uncertainty and bias in fractional cover estimates. This uncertainty is greater when the difference between the pixel size of the image and a plot size for estimating cover is small. Despite these challenges, remote sensing is the only feasible approach for providing spatially comprehensive cover estimates in a cost-effective manner for sustainable long-term community monitoring.

Spatial Scale for Estimating Life-form Cover

Whether estimating fractional cover in the field or using remote sensing, a basic methodological issue that must be addressed in order to establish monitoring protocols is identifying an appropriate spatial scale or plot size of observation. Each ecosystem process and function has an inherent spatial and temporal scale, and data should be collected and analyzed at appropriate scales (Levin, 1992). Fractional cover should be estimated at a suitable scale for ecologically meaningful analysis and quantification of ecological integrity. However, determining such a spatial scale for quantifying cover is challenging. Bakker and others (1996) state "…we do not know what the optimal [plot] size is. It depends on the hypotheses to be tested, the vegetation type involved and financial and technical constraints as to the maximal duration of the research (p.147)."

In general, the size of plants dictates an appropriate plot size (Mueller-Dombois and Ellenberg, 1974) so that a plot contains multiple individuals and represents a sample of community diversity. Commonly a 1 m^2 plot is utilized for herbaceous vegetation while plot sizes ranging from 25 m^2 to 1,000 m^2 and from 100 m^2 to 10,000 m^2 are often used to sample shrubs and trees, respectively (Mueller-Dombois and Ellenberg, 1974; Whittaker, 1977). Although there is no scientifically supported single plot size for estimating fractional cover in all plant communities, 20 m x 50 m plots have been widely used for studying a range of community types (Peet, 1981; Stephens et al., 2009; Stohlgren et al., 1995). Therefore, this plot size may be considered desirable for comparative reasons. Sub-plots may be nested within a larger plot, which allows researchers to obtain data at multi-scales from 1 m^2, 10 m^2, 100 m^2, to 1,000 m^2 (Figure 4). This 'tenth-hectare plot' was originally designed for studying plant diversity (Stohlgren et al., 1995; Whittaker, 1977), and species' cover is often one of the sampling variables. In contrast, smaller square plots (10 m x 10 m) were used by Parsons (1976) to estimate cover-abundance of woody species (and to record height) in order to examine vegetation structure in scrub communities of California and Chile.

The selection of spatial scale is important particularly for spatially heterogeneous areas because scale effects increase with community heterogeneity (Fortin, 1999). Deutschman (2008) evaluated life-form-level fractional cover, as well as dominant species cover and species richness, as monitoring variables using 20 m x 20 m and 50 m x 20 m plots. The size of plot was determined to be the least critical factor affecting estimates of fractional cover for functional groups (and species) among factors examined (i.e. sampling locations, plots, methods and observers); there was little variability in cover estimates between the plot sizes tested in the study. However, Schlup and Wagner (2008) reported that the spatial scale of sampling can substantially impact the variance of cover and spatial pattern

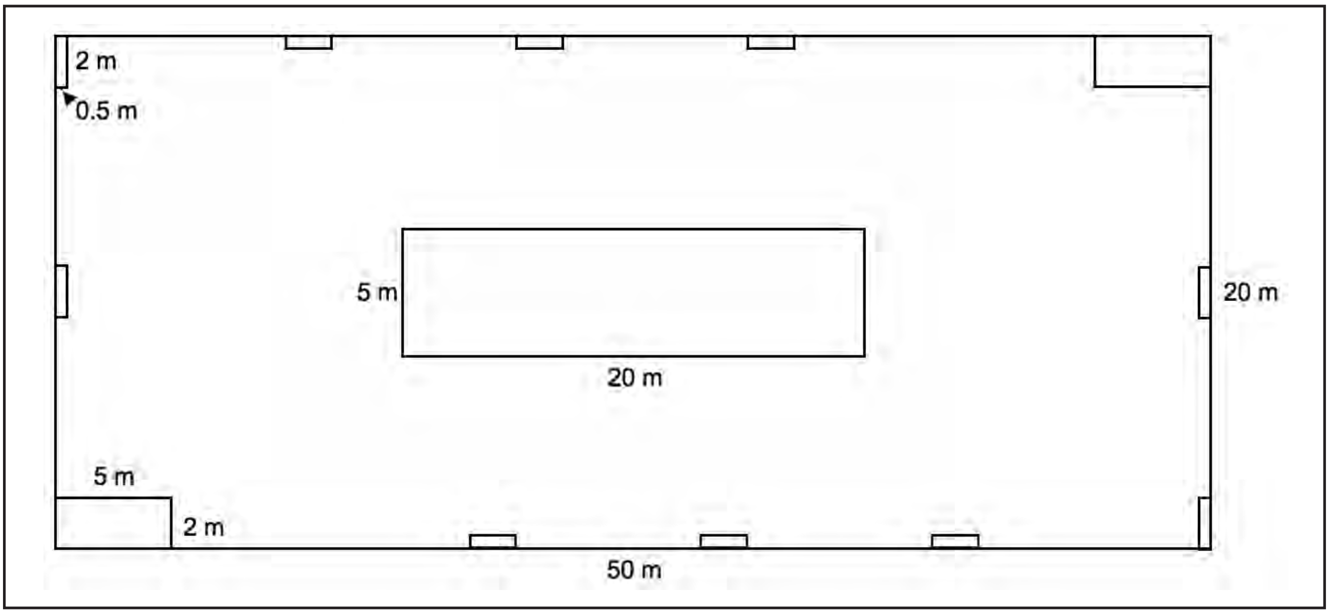

Figure 4 Modified-Whittaker sampling design. A tenth hectare (20 m x 50 m) plot contains one 100 m^2 plot at the center, two 10 m^2 subplots at upper right and lower left corners, and ten 1 m^2 subplots (0.5 m x 2 m) along the edge of the plot. Modified from Stohlgren et al. (1995).

measures such as spatial autocorrelation. Duncan et al. (1993) reported increased variability in cover estimated within 60 m x 60 m plots compared to estimates from 100 m x 100 m plots because of heterogeneity within a community.

An appropriate plot size may be determined through systematic sampling approaches. Because it is expensive and time consuming, exploring appropriate plot sizes based on field-based sampling has been limited. Utilizing remotely sensed imagery and spatial analysis tools (e.g. semivariogram and spatial statistics), effects of plot size on fractional cover estimates could be examined, which would provide insights for selecting an appropriate spatial unit for cover estimates. From a spatial statistical perspective, a potential criterion for selecting an ideal spatial scale is the maximum extent with the minimum variability in cover fraction within a plot or sampling unit (Atkinson, 1997; Curran and Atkinson, 1998). Otypkova and Chytry (2006) noted that the plot size should be small enough to contain little variation and large enough to reflect typical conditions of the community. Therefore, the optimal plot size could be a function of homogeneity and representativeness. Phinn and others (1996) explored appropriate spatial scales for sampling vegetation within shrub-invaded grassland communities in the Chihuahuan Desert by analyzing spatial patterns of a spectral vegetation index derived from high resolution remotely sensed imagery using semivariograms. They found that the sampling interval of 5 m (with 1 m pixel size) is optimal for estimating above ground biomass for high desert grassland and shrubland community types.

A single standardized plot size within a community is generally desirable (Chytry and Otypkova, 2003). For heterogeneous, complex, and dynamic communities, however, identifying a single appropriate plot size for monitoring cover of multiple cover types (e.g. shrub, herb, and bare ground) is unlikely. One of the reasons is that each cover type exhibits unique spatial variation and pattern. In many arid and semi-arid communities, shrubs typically form clusters and herbaceous vegetation and bare ground fill in gaps between those clusters. A degraded shrubland could have relatively homogeneous herbaceous cover. For bare ground, soil patches appear as linear features along trails, while rock outcrops tend to be clustered. This variation also imposes a challenge for determining a single spatial unit for estimating fractional cover of the cover types over time. For example, a community at an intermediate stage of succession may have more heterogeneous spatial structure than a community during the first few years following fire, when herbaceous cover is quite high. Mosaics of various age stands within a community created by multiple disturbance regimes and topography make spatial structure even more complex.

Effectiveness of Cover for Monitoring Arid and Semi-arid Communities

Fractional cover has been a crucial variable for monitoring conditions of rangelands and shrublands in interior west and southwestern U.S. (Browning et al., 2008; Henkin et al., 1998; Wilson and Tupper, 1982). Change in proportions of shrubs, herbaceous plants, and exposed soil influences rangeland productivity, indigenous ecosystem function, and biochemical cycles.

From the perspective of rangeland productivity, community health is primarily quantified based on how well an ecosystem conserves and maintains its soil and water resources (Herrick and Whitford, 1995). A combination of soil erosion rates and productivity (or vegetation cover) has been utilized as a proxy of rangeland health. Life-form cover is incorporated in several indictors of rangeland health, which were originally developed for the Chihuahuan Desert (de Soyza et al., 1998; Whitford et al., 1998) and later applied to rangelands in the Great Basin (de Soyza et al., 2000). De Soyza and others (2000) suggest that the increase in herbaceous cover indicates recovery from recent disturbance and thus, decreased risk of soil erosion. This can be translated as improved rangeland health. They also noted that bare soil cover (or patch size) directly indicates soil erosion rates, and vegetation cover consisting of shrubs and herbs can be used for an earlier indicator of rangeland degradation.

From the community conservation perspective, community health is defined by the sustainability of native ecosystem function such as sufficient cover of native shrub and absence of non-native plants (King 1993, Rapport et al. 1998). Non-native grass invasion and woodland encroachment are major threats to the sagebrush-steppe community type of the Great Basin (Wisdom and Chambers, 2009). Cover is the primary variable used for assessing extent or risk of community-type conversion. For example, shrublands associated with *Artemisia* spp. provide habitat to a threatened bird species, *Centrocercus urophasianus* (greater sage-grouse), and the habitat has dramatically declined since European settlement (Archer, 1994; Homer et al., 2009). Monitoring overall shrub cover has been playing an important role in habitat conservation because shrublands in the Great Basin are often dominated by native *Artemisia* spp. (sagebrush). Fractional cover maps have been providing information about species-specific habitat requirements for ecological modeling particularly in Wyoming, Utah, and Montana (Aldridge et al., 2008; Doherty et al., 2003; Homer et al., 1993).

Life-form cover, particularly shrub cover, is also recognized as an important indicator of ecosystem integrity of rangelands in the American Southwest, where shrub encroachment into native grasslands and dunelands is of primary concern (Archer, 1994). For nearly a half century, tracking change in shrub cover, which is dominated by *Prosopis glandulosa* (honey mesquite), has been the focus of ecosystem management in the region (Buffington and Herbel, 1965; Hennessy et al., 1983; Gibbens et al., 2005). Ansley and others (2001) estimated shrub encroachment rates in northern Texas between 1976 and 1995 based on changes in shrub cover in grassland communities in order to determine economically feasible treatments.

Remote sensing is an indispensible tool for estimating fractional cover in the Intermountain West of the U.S. because of the vast area of rangelands and shrublands in the regions. A number of studies demonstrate the utility of remote sensing for estimating fractional cover in rangelands (Laliberte et al., 2004; Seefeldt and Booth, 2006; Homer et al., 2008). Ramsey and other (2004) mapped life-form cover (i.e. tree, shrub, grass, and forb) within shrub-steppe environments in south-central Utah using 30 m spatial resolution Landsat ETM+ multispectral imagery. Observed fractional cover in the field and cover estimated based on reflectance values and normalized vegetation index (NDVI) were highly correlated ($r^2 > 0.75$), particularly for shrub and herbaceous (i.e. grass-forb combined) cover in sagebrush-dominated communities. Homer and others (2008) mapped fractional cover of shrub, herb, litter, and bare ground of entire State of Wyoming by developing a model using field measurements and multi-resolution remotely sensed imagery. The estimation accuracy based on RMSE ranges from approximately 3 to 6% for shrub and 6 to 12% for bare ground.

In rangelands in the Southwest, Laliberte and others (2004) generated time-series maps of plant life-form cover within a rangeland of southern New Mexico from 1937 to 2003 using aerial photography and high-resolution remotely sensed imagery in conjunction with an image segmentation approach known as object-based image analysis. The study demonstrated the effectiveness of advanced image processing techniques for identifying the trajectory of grassland-to-shrubland conversion by comparing change in life-form cover proportion over time, as well as change in area occupied by life forms. Browning and others (2008) mapped change in shrub cover and its location in the Sonoran Desert from 1936 to 1996 using 1-m resolution scanned color infrared aerial photography in order to understand the trend of communities and potential causes of the trend. These studies strongly suggest that the utility of remote sensing for estimating cover in arid and semi-arid vegetation communities.

Incorporating Life-form Cover for CSS Community Monitoring

As discussed earlier, vegetation type mapping and field sampling of individual plants are two methods currently employed for monitoring CSS communities. For San Diego County, a vegetation type map at community-type and association levels (Holland, 1986) was first created in 1995 in digital format and has been updated based on image interpretation of aerial photography (SANDAG, 2004). A new initiative is underway for re-mapping vegetation community types based on a standardized classification scheme adopted by State of California and U.S. federal agencies (Sawyer and Keeler-Wolf, 1998). The City of San Diego (2008) has conducted plot-level field surveys of rare plant species since 1999, though monitoring variables and field sampling methods have varied. In the community monitoring plan in Orange County, California, fractional cover of dominant and sub-dominant species, percent of native and non-native species, relative species cover, and diversity have been incorporated in monitoring protocols in order to establish baselines for detecting early signs of adverse change of CSS communities such as type conversion (OCPW, 2006). Both vegetation type maps and field-based data provide ecologically meaningful information such as distributions of plant community across space and species richness within sampling plots. However, there is a need for monitoring variables which are spatially comprehensive yet describe internal conditions of a community by characterizing vertical and horizontal structures within a community.

The effectiveness of plant life-form cover in semi-arid interior west and southwestern U.S. that was discussed above suggests that fractional cover of plant life-form is more likely to be an ecologically meaningful monitoring variable for CSS communities. In fact, resource managers and scientists have found that cover is a useful monitoring variable for biological conservation of the community type. Westman and O'Leary (1986) consider percent cover as one of the most effective variables for quantifying resilience of CSS communities after fire. A team of experts (e.g. habitat preserve managers, wildlife ecologists, and habitat conservation specialists) has identified that cover of true shrub, subshrub, and bare ground are the key indicators of CSS habitat condition along with abundance of invasive species (Coulter et al., 2003). Cover of native shrub and other native life-forms have been identified as useful variables in the conceptual model for CSS community monitoring (Hierl et al., 2007). Increasing herbaceous cover is often indicative of exotic species invasion, which suggests degradation of CSS communities, while increasing shrub cover is indicative of increased resistance to exotic invasion because reduced light penetration has negative affect on undergrowth (Keeley et al., 2005; Westman and O'Leary, 1986).

Similar to rangelands in the Intermountain West of the U.S., cover would be a valuable measure for assessing species-specific habitat quality in CSS communities, which is the main objective for many conservation programs. Cover indicates the amount of canopy and bare ground exposure that may be key requirements for foraging and shelter for vertebrate species. For example, an endangered bird, *Polioptila californica* (California gnatcatcher), prefers CSS communities that have average shrub cover of 50% (Beyers and Wirtz, 1995) with exotic cover no greater than 40% (Hunsaker et al., 2000). A lizard species protected by the state of California, *Cnemidophorus hyperythrus* (orange-throated whiptail), requires approximately 10 to 40% bare ground exposure within CSS habitat for foraging and thermoregulation (McGurty, 1981). An endangered rodent, *Dipodomys stephensi* (Stephens' kangaroo rat), prefers habitats with abundant bare ground and herbaceous cover (e.g. approximately 70 to 90%) (Goldingay and Price, 1997).

Although CSS communities are not as extensive as typical rangelands, remote sensing would be the appropriate tool for estimating life-form fractional cover. The CSS community type, which experiences a recurring disturbance-succession cycle, more likely require frequent monitoring due to its high spatial heterogeneity and temporal dynamics. Thus, field-based approaches would be ineffective for sustainable long-term ecological monitoring. Witztum and Stow (2004) tested remote sensing approaches for estimating bare ground cover in CSS communities of southern California and found strong correspondence between the level of habitat degradation caused by recreational activities and bare ground fraction estimated using remotely sensed imagery. Spatially exhaustive data acquisition and the synoptic view of remote sensing permit data collection from inaccessible areas, which is a great benefit for CSS community monitoring because:

- CSS communities often occur in areas of complex terrain and/or are surrounded by an environment that limits physical access (e.g. no access roads or trails and surrounded by dense chaparral stands);
- CSS supports a large number of rare species and more than 70% of endangered species rely on private lands (Luoma, 1998); and
- Sensitive habitat that has experienced recent perturbation (e.g. fire, grazing, and heavy recreational activities) should be protected from further disturbance (e.g. tramping) for recovery/rehabilitation.

In California's Mediterranean-type ecosystems, various spatial scales have been utilized for studying vegetation communities including Vegetation Type Map (VTM) plots (i.e. 10 m x 40 m) for examining vegetation trends (Franklin et al., 2004; Keeler-Wolf, 2007; Wieslander, 1935) and 20 m x 20 m quadrats for examining historical decline of CSS communities in the Riverside-Perris Plain (Minnich and Dezzani, 1998). Fractional cover estimated from 25 m x 25 m plots has been utilized to study resilience, and the plot size has been most frequently utilized to estimate fractional cover within CSS communities. Westman and O'Leary (1986) estimated percent cover of true shrubs and subshrubs (and recorded floristic composition) within a CSS community in the western Santa Monica Mountains before and after a fire event over six years to assess resilience of the CSS community to fire. They identified vigor of resprouting shrubs as a good indicator of competitive success following disturbance. O'Leary and Westman (1988) utilized the same or similar plot data to examine disturbance effects on succession pattern in the early post-fire period and found that weak shrub resprouting contributed to herbaceous growth. The same 25 m x 25 m plot size was utilized to estimate fractional cover of dominant shrubs in order to examined influence of fire frequency on CSS community composition (Malanson, 1984). Westman (1981) also utilized fractional cover estimates from plots of this size to analyze changes in diversity with post-fire succession. He noted importance of areal coverage to account for spatial heterogeneity in order to inform conservation management. Because of the precedent set by a large number of scientific studies conducted on CSS community sustainability or resilience, a 25 m x 25 m spatial sampling unit may be considered appropriate for estimating fractional cover within CSS communities for comparative reasons.

Diffendorfer and others (2007) utilized 50 m x 50 m sites to estimate fractional cover at a life-form level to explore useful variables for the Index of Biological Integrity (IBI; Karr 1981), a single numeric value indicating the state of a community, specific to the CSS community type. This plot size was selected because it was deemed to correspond to the maximum extent for planting, exotic plant eradication, or erosion control in a day to a week. Thus, a 50 m x 50 m spatial scale is a reasonable spatial scale for estimating life-form cover from the management perspective.

Conclusions and Research Needs

Monitoring ecological variables that provide information about community integrity at an appropriate spatial scale is vital to conserving these biologically diverse CSS communities. Comprehensive monitoring requires measures of multiple indicators at multiple scales in order to adequately describe the state of communities (Andreasen et al., 2001; Woodley and Theberge, 1992). While coarse-scale, spatially-comprehensive (i.e. vegetation type maps) and fine-scale-site-specific (i.e. field) data provide ecologically valuable information about CSS communities, there is an information gap between these data sources. Fractional cover at a life-form level has potential to compensate for shortcoming of existing CSS monitoring methods.

Life-form fractional cover is a valuable measure and has been widely incorporated into monitoring programs for shrublands and physiognomically comparable rangelands. A number of studies and reports for the CSS community type recognize that life-form cover as a useful monitoring variable to assess habitat sustainability, which is the key component of CSS community integrity. Biologists and local governments have been exploring field-based approaches for cover estimates; however, results are not yet conclusive and sampling schemes are inconsistent. Remote sensing is the only feasible approach for obtaining spatially comprehensive, cost-effective information over large areas. The use of remote sensing does not eliminate the need for either field measurement or vegetation type mapping; rather it will likely require a combination of approaches to reliably estimate life-form cover and provide comprehensive information for long-term monitoring of CSS integrity. Investigation of the reliability, robustness, and efficiency of remote sensing approaches for cover estimate within shrub-dominated vegetation communities is also required in order to understand full range of utility of remote sensing for quantifying and monitoring ecological integrity of such ecosystems.

Although there is no single well-established, scientifically supported spatial scale for estimating fractional cover, a 25 m x 25 m square plot size would be the most appropriate scale because the plot size has been commonly employed for studying resilience of CSS communities and for habitat management. A previous study also suggests that cover estimates are not particularly sensitive to plot size within a range that encompasses this plot size. Definitive prescription of appropriate plots sizes requires further investigation of scale effects on fractional cover over a greater spatial range using remote sensing and spatial analysis tools.

Our review and synthesis of information about plant life-form cover in arid and semi-arid vegetation communities reveal that life-form fractional cover would provide ecologically meaningful intermediate-scale information, which is unattainable from vegetation type maps and species-level field data for CSS community. Thus, we strongly recommend incorporating fractional cover of true shrub, subshrub, herb, and bare ground in monitoring variables for CSS community conservation. Estimating life-form cover at a 25 m x 25 m spatial scale using remote sensing in conjunction with field validation would be the most reasonable approach for initial implementation. Investigation of remote sensing techniques and an appropriate spatial scale, collaboration of experts and scientists, and refinement of protocols are essential for fully integrating life-form fractional cover estimation into strategies for sustainable long-term monitoring of CSS communities. These concepts and framework are not limited to CSS communities of southern California. They can also be applied for developing/refining monitoring strategies for other shrub communities in Mediterranean-type ecosystems and physiognomically comparable vegetation communities in the world.

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