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Structural diversity in rangelands: a framework for quantifying what makes a functional rangeland

Olsoy, PJ[1]; Copeland, SM[1]; Caughlin, TT[2]; Duquette, CA[3]; O'Connor, RC[1]; Boyd, CS[1]

[1] Eastern Oregon Agricultural Research Center, USDA Agricultural Research Service, Burns, OR 97720; [2] Department of Biological Sciences, Boise State University, Boise, ID 83704; [3] Eastern Oregon Agricultural Research Center, The Nature Conservancy, Burns, OR 97720

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Abstract

Rangelands across the globe are threatened by factors such as climate change, altered fire regimes, and annual grass invasion, often leading to simplified vegetation structure and reduced ecosystem function. Restoring degraded rangelands to their original state is not always possible given socially acceptable levels of financial and capacity commitment. Whether the goal is to reestablish historically occurring flora and fauna or to mitigate some of the negative impacts of a degraded system, managing and restoring these ecosystems requires knowledge of what makes rangelands multifunctional systems (e.g., grazing, wildlife habitat, recreation) and what causes declines in these ecological functions following degradation. Structural diversity metrics can be used as an indicator of ecosystem function and are now possible to continuously measure across landscapes with remote sensing. Recently, the use of structural diversity from 3-dimensional (3D) spatial datasets has been proposed as a flexible method to measure ecological functions in forested systems but has yet to be applied to rangeland management. We propose using structural diversity to monitor rangeland ecosystem function with two case studies. First, we measure structural diversity across a series of ecological states in semiarid rangelands, from intact shrub and native bunchgrass communities to invasive annual grass-dominated sites and multiple phases of juniper (*Juniperus* spp.) encroachment. Second, we compare structural diversity between paired grazed and ungrazed landscapes. We found that structural diversity differs across ecological states, demonstrating a potential way to assess ecosystem function. With the recent increase in the availability of high-resolution 3D structural data from low-cost unoccupied aerial systems (UAS), structural diversity could be used to help managers rapidly assess the ecological function of rangelands.

Introduction

Recent advances in technologies, such as more widely available LiDAR and low-cost unoccupied aerial systems (UAS or drones) bring new opportunities to understand structure—function relationships (Anderon and Gaston 2013). Researchers in forested ecosystems have begun to explore these relationships, suggesting a theoretical framework focusing on the vertical structuring and niche partitioning seen in these systems (Atkins et al. 2018; LaRue et al. 2023), and linking some of these metrics to ecosystem functions such as productivity (LaRue et al.

2019), but little work has been done in rangelands (see Zaiats et al. 2024), despite demonstrated linkages between structural heterogeneity and ecosystem function using other methods (Maestre et al. 2016). Besides estimating structural contributions to biodiversity, structural diversity could assess fire resilience (e.g., patch arrangement of woody fuels), recovery from disturbances (e.g., post-fire, agricultural abandonment), wildlife habitat, and management treatment longevity. Compared to forested ecosystems, rangelands have less vertical stratification, and therefore different metrics and spatial scales should be considered.

In this study, we provide two case studies demonstrating a framework (Fig. 1) for applying structural diversity metrics to management of rangelands: 1) three ecological states representing intact shrub steppe, invasive annual grass invasion, and juniper encroachment; and 2) a long-term grazing exclosure study (88 years) comparing six sets of paired grazed and ungrazed pastures.

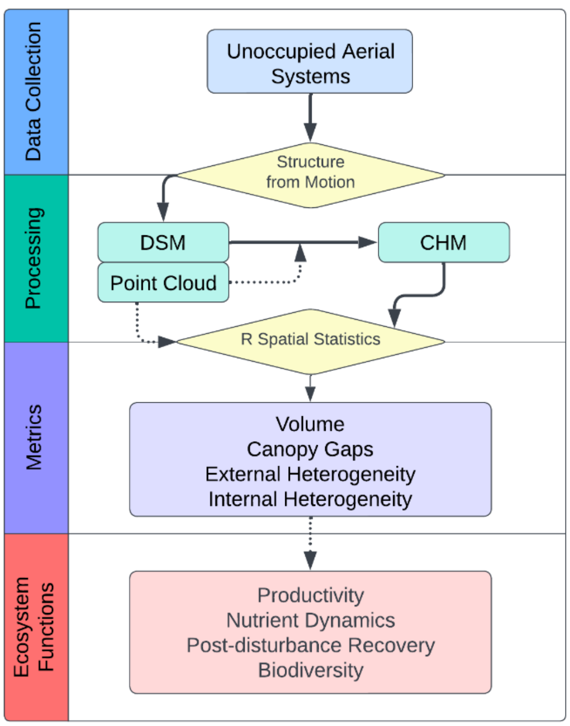

Figure 1. Workflow for generating structural diversity metrics with unoccupied aerial systems (UAS) structure from motion to generate digital surface models (DSM) and canopy height models (CHM).

Methods

We conducted our research at two study areas in southeastern Oregon, USA. The first case study was in the Stinkingwater Mountains (43.63°N, 118.38°W), which contains a range of vegetation communities and ecological states from intact shrub steppe dominated by *Artemisia tridentata*, to post-fire landscapes invaded by medusahead (*Taeniatherum caput-medusae*), to conifer (*Juniperus* sp.) encroached shrublands. The second case study was a long-term grazing exclosure study at the Northern Great Basin Experimental Range (NGBER, 43.48°N, 119.71°W), with pastures covering a range of elevations and vegetation communities with *A. tridentata* dominating the lower elevations and *J. occidentalis* woodlands dominating the higher elevations, but little annual grass invasion compared to the Stinkingwater Mountains.

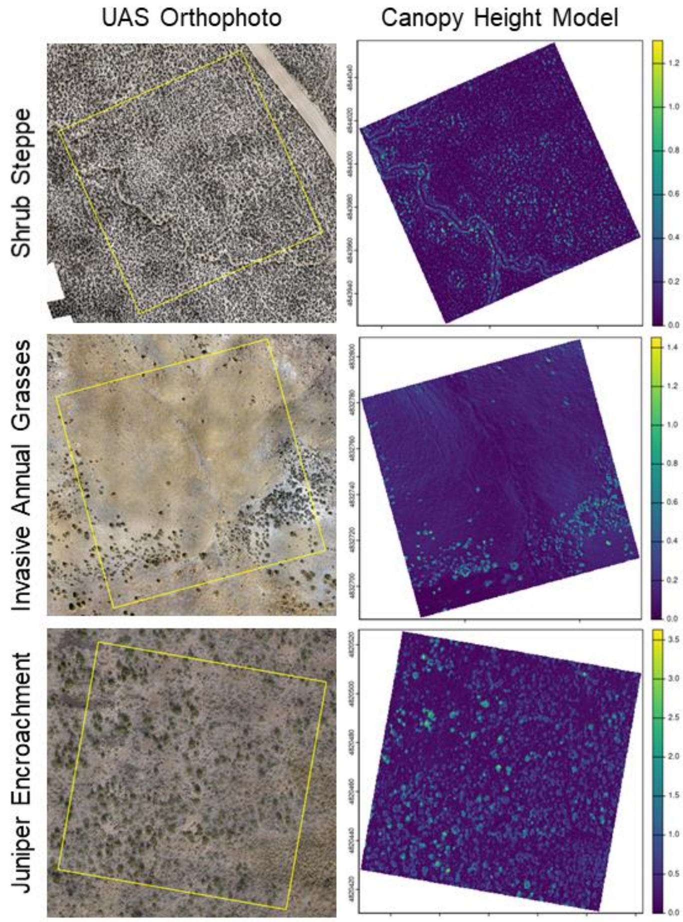

Figure 2. Unoccupied aerial system (UAS) high-resolution orthophotos and canopy height models for three sites in Oregon, USA representing healthy shrub steppe, annual grass invaded, and juniper encroached ecological states. Yellow boxes represent the 100 x 100 m (1 ha) assessed in this study.

We collected imagery with a Freefly Astro (Freefly Systems, Woodinville, WA, USA) UAS equipped with a 61 MP camera. We flew the UAS at 40 m altitude above ground level, with a nadir flight and a 30-degree offset flight with a cross-grid pattern resulting in ~1-cm pixel resolution. We processed the UAS imagery with Open Drone Map (ODM version 3.3) on USDA's SCINet supercomputer Atlas with 48 cores and 320 GB of RAM. ODM parameters were feature-quality and pc-quality set to 'ultra,' min-num-features of 40,000, and orthophoto-resolution and dem-resolution set to 0.01 to obtain the highest pixel resolution from the data. We generated digital surface models (DSM) using structure from motion photogrammetry (Cunliffe et al. 2016; Olsoy et al. 2018). To standardize our structural diversity metric comparisons, we clipped a 100 x 100 m (1-ha) region out of each image for testing purposes. We generated a canopy height model (CHM) by subtracting a ground surface (minimum height on a 1-m moving window) from the digital surface model generated by ODM (Fig. 2). We calculated structural diversity metrics related to volume, openness (gaps), and heterogeneity (internal and external) (LaRue et al. 2019) at a 1-m spatial scale in R version 4.4 (R Core Team 2024) with the terra package (Hijmans 2024). Volume was summed across the 1-ha plot. We calculated the percent of canopy gaps (pixels with less than 15 cm vegetation height) aggregated to 1 m and then averaged across the 1-ha plot. Heterogeneity was calculated as the standard deviation of vegetation heights, with external structural heterogeneity representing the average standard deviation across the 1-ha plot, and internal structural heterogeneity calculated as the standard deviation of the standard deviation of height across the 1-ha plot (LaRue et al. 2019). For the grazed-ungrazed sites (n = 6 pairs), we used paired t-tests to assess differences in structural diversity metrics.

Results

Ecostates

Unsurprisingly, volume was highest in the juniper site with 3610 m^3ha^{-1} compared to both the shrub steppe site (1046 m^3ha^{-1}) and the invasive annual grass site (1480 m^3ha^{-1}). Structural heterogeneity metrics were lowest in the invasive annual grass site (external = 0.046, internal = 0.055) compared to the shrub steppe site (external = 0.096, internal = 0.072) and the juniper encroached site (external = 0.201, internal = 0.151).

Grazing

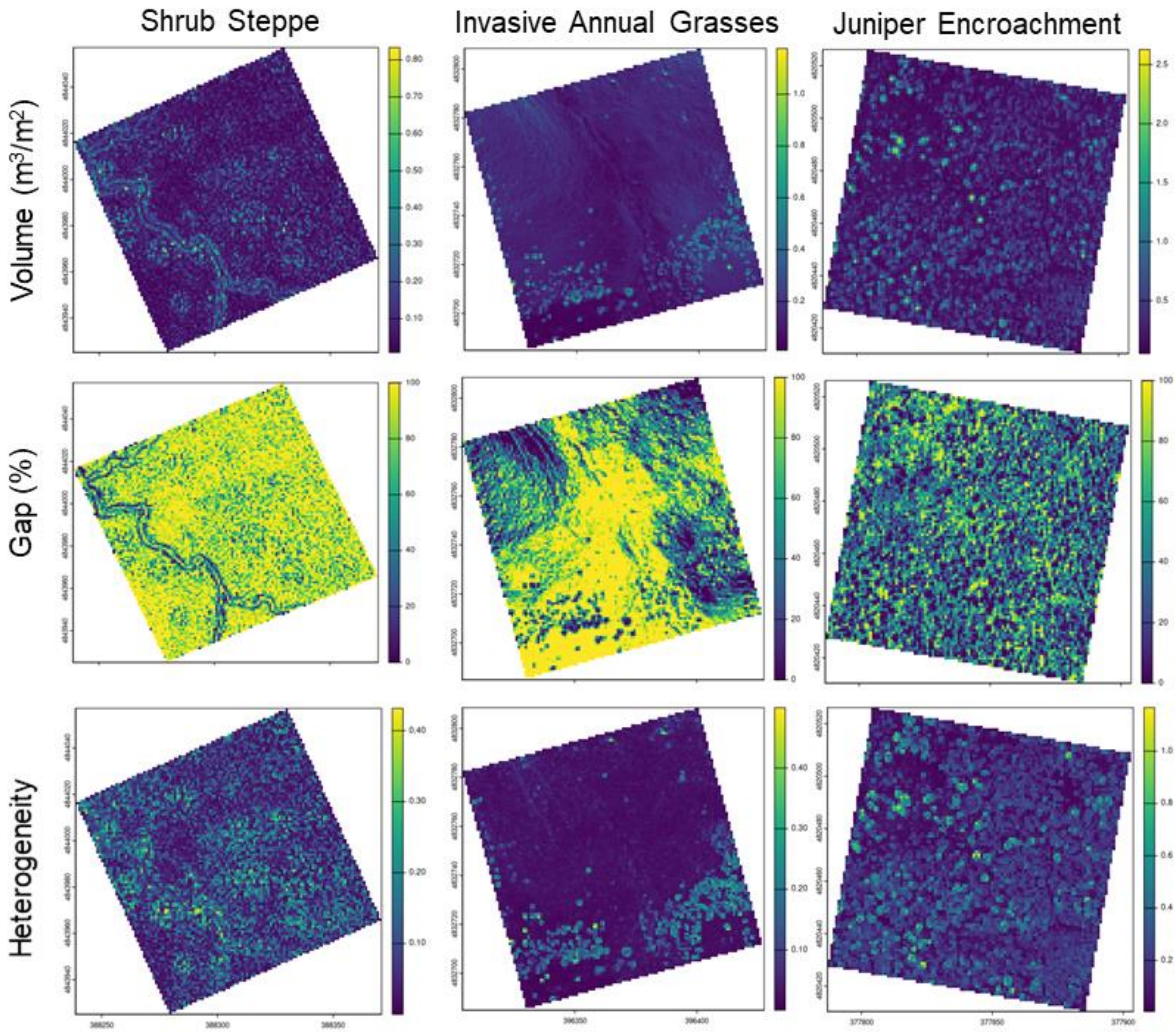

Figure 3. Structural diversity metrics for three sites representing distinct ecological states in Oregon, USA.

We did not find any statistical differences in structural diversity metrics between grazed an ungrazed plots at NGBER. Volume was 4508 m^3ha^{-1} in grazed and 4690 m^3ha^{-1} in ungrazed plots (t_5 = -0.36, P = 0.733). Gap percent was marginally significant with 56% gaps in grazed and 49% in ungrazed plots (t_5 = 2.45, P = 0.058). External heterogeneity was 0.19 in both grazed and ungrazed plots (t_5 = -0.05, P=0.96) and internal heterogeneity was 0.23 in both grazed and ungrazed plots (t_5 = -1.21, P = 0.28). Variability was higher between sites than treatments, for example, the site with the most juniper had 15,458 m^3 volume, while the non-juniper pastures had between 1147 and 1767 m^3 volume. Percent gaps also ranged from 0.1% at the juniper site to over 80% at two of the grazed non-juniper sites.

Discussion

Structural diversity metrics differed across ecological states, with juniper encroached sites having higher volume and less canopy gaps. However, we did not detect structural diversity differences in grazed pastures compared to long-term grazing exclosures, perhaps due to the small sample size (n = 6). The light to moderate grazing at the site could explain our results, which did not lead to differences in species composition (Copeland et al. 2021). The scale we used to calculate structural diversity metrics (1 m) could also be wrong for the ecological process (Levin 2000). The relative contribution of different plant functional types and species to structural diversity is unknown and could shed light on differences in biodiversity and occupied niches, which are undetected when looking at site-level structural diversity metrics. For example, the volume and canopy connectivity seen at the invasive annual grass invaded site is due to large mats of medusahead and indicate high wildfire fuel loading within a degraded system, while similar volume and connectivity in the shrub steppe ecological state represents high shrub cover with less connectivity of fine fuels. Selecting metrics that quantify these differences in wildfire risk and biodiversity is vital (Levin 2000; Ellsworth et al. 2020). Rangelands likely have differences in useful spatial scales (Zaiats et al. 2024) and less emphasis on vertical structural diversity than forests, particularly semiarid rangelands which tend to have lower productivity. However, more productive rangelands such as African savannas or the central plains of USA could demonstrate some of the vertical stratification and utilize the wealth of structural diversity metrics already developed for forested systems.

Future research could explore how metrics change across different scales and seek to directly link structural diversity metrics with ecosystem functions. A few examples of future work are landscape prioritization for fuels management, assessing resistance and resilience after disturbances, and developing workflows and tools to ease the implementation of these across broad landscapes and ease administrative burdens for land management agencies.

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