California Agriculture

April–September 2022 | Volume 76 Number 2–3

University of California | Peer-reviewed Research and News in Agricultural, Natural and Human Resources

California Agriculture

Peer-reviewed research and news published by University of California Agriculture and Natural Resources

ORDERS AND SUBSCRIPTIONS

2801 Second Street, Room 181A; Davis, CA 95618-7779
Phone: 530-750-1223; Fax: 530-756-1079; calag@ucanr.edu

EDITORIAL

2801 Second Street, Room 184; Davis, CA 95618-7779
530-750-1223; calag.ucanr.edu

California Agriculture (ISSN 0008-0845, print, linking; ISSN 2160-8091, online) is published quarterly. Postmaster: Send change of address "Form 3579" to *California Agriculture* at the address above.

California Agriculture is a quarterly, open-access, peer-reviewed research journal. It has been published continuously since 1946 by University of California Agriculture and Natural Resources (UC ANR). There are about 3,000 print subscribers.

Mission and audience. *California Agriculture* publishes original research and news in a form accessible to an educated but non-specialist audience. In the last readership survey, 33% of subscribers worked in agriculture, 31% were university faculty or research scientists and 19% worked in government.

Electronic version of record. In July 2011, the electronic journal (calag.ucanr.edu) became the version of record. Since then, some research article are published online only. All articles published since 1946 are freely available in the online archive, calag.ucanr.edu/Archive/.

Indexing. The journal provides article metadata to major indexing services, including Thomson (Web of Science), AGRICOLA, the Directory of Open Access Journals and EBSCO (Academic Search Complete), and has high visibility on Google Scholar. All articles are posted to eScholarship, UC's open-access repository. In the 2020 Thomson JCR, the journal's impact factor was 1.64.

Authors. Most authors (75%) are among the roughly 1,000 academics affiliated with UC ANR, including UC Cooperative Extension specialists, advisors and academic coordinators; and faculty in the following UC colleges: UC Berkeley College of Natural Resources, UC Davis College of Agriculture and Environmental Sciences, UC Davis School of Veterinary Medicine, and UC Riverside College of Natural and Agricultural Sciences. Submissions are welcome from researchers based at government agencies and at other campuses and research institutes.

Article submission and review. Guidelines for authors are here: calag.ucanr.edu/submitarticles/. The journal uses a double-blind peer-review process described at calag.ucanr.edu/About/. Roughly 50% of all submissions are rejected by the editors without peer review due to a mismatch with the journal's scope or clear weaknesses in the research. Of the subset of submissions that enter the peer-review process, roughly 60% are ultimately accepted. All accepted manuscripts are edited to ensure readability for a non-specialist audience.

Letters. The editorial staff welcomes letters, comments and suggestions. Please write to us at the address below, providing your contact information.

Print subscriptions. These are free within the United States. Go to: calag.ucanr.edu/subscribe/ or write or call.

UNIVERSITY OF CALIFORNIA

UNIVERSITY OF CALIFORNIA
Agriculture and Natural Resources

CONTENTS

APRIL–SEPTEMBER 2022 • VOLUME 76, NUMBER 2–3

Research and review articles

COVER: Due to ongoing climate change, widespread wildfires are predicted to increase over time in California. New research findings suggest that land managers can use strategic livestock grazing to reduce fuel loads in grasslands, which could help lower fire hazards (see Ratcliff et al., page 60). Photo credit: © Andreistanescu, Dreamstime.com.

RESEARCH ARTICLE

Cattle grazing reduces fuel and leads to more manageable fire behavior

Grazing cattle can help reduce fuel loads on rangelands and mitigate the ever-growing risk of catastrophic wildfires.

by Felix Ratcliff, Devii Rao, Sheila Barry, Shane Dewees, Luke Macaulay, Royce Larsen, Matthew Shapero, Rowan Peterson, Max Moritz and Larry Forero

Online: https://doi.org/10.3733/ca.2022a0011

Abstract

Cattle play an important role in wildfire management by grazing fuel on California rangelands. The benefits of cattle grazing have not been thoroughly explored, though. Using statewide cattle inventory, brand inspection and land use data, we have estimated that cattle removed 11.6 billion pounds (5.3 billion kilograms [kg]) of non-woody plant material from California's rangelands in 2017. Regionally, these reductions varied between 174 and 1,020 pounds per grazed acre (195 to 1,143 kg per hectare). Fire behavior is characterized in this paper by flame length. Fire behavior models suggest that these regional fuel reductions lower flame lengths, and lead to more manageable wildfires. In addition, fire-based models show that cattle grazing reduces fuel loads enough to lessen fire hazards in many grazed areas. Moving forward, there may be significant opportunities to expand strategic grazing on rangelands to add extra layers of protection against wildfires.

Recent wildfire seasons in California have been some of the worst on record. This "new reality" highlights the importance of understanding how land management practices such as cattle grazing affect wildfire behavior. Fire behavior is characterized in this paper by flame length. While climate change can lead to more severe fire behavior for California wildfires, our findings suggest that land managers can help balance out these dangers in grasslands by using livestock grazing to reduce fuel loads. CAL FIRE's California Vegetation Treatment Program (CalVTP) utilizes prescribed herbivory, which is the targeted grazing of cattle, sheep and goats to reduce wildland plant populations. While not included in CalVTP, conventional grazing also plays an important role in fuel load reductions.

Livestock grazing is a prevalent land use on California's rangelands and is considered a cost-effective method of reducing fuel loads (Taylor 2006). As such, fuel reduction through livestock grazing is a

These stocker cattle graze seasonally, during spring, reducing fine fuels across a large landscape. *Photo:* Devii Rao.

common management goal in regional, state, county and agency management plans (EBMUD 2000; EBRPD 2013; George and McDougald 2010; Rancho Mission Viejo 2006; Santa Clara County Parks 2018). However, management plans generally do not list target fuel conditions to achieve through livestock grazing.

Since livestock grazing is already in widespread use for wildfire fuel management in California, it is important to understand in greater detail to what extent livestock reduce fuel loads across the state, including how this varies spatially. More research on grazing for fuel reduction has been done on sheep and goats than on cattle (Nader et al. 2007). Especially in California, much of this research has focused on forests and shrublands rather than grasslands, and on woody rather than herbaceous fuels (Green and Newell 1982; Minnich 1982; Narvaez 2007; Tsiouvaras et al. 1989). While cattle graze all rangeland types in California, they primarily graze grasslands, preferring herbaceous forage like grasses and flowering plants (Launchbaugh et al. 2006; Van Soest 1994). When these fuels dry out, they are known as “fine fuel” — fuels with a high surface-area-to-volume ratio that can be quickly combusted in wildfires (USFS 2022). Because they are by far the most widespread and abundant domestic grazers in the state (Saitone 2018), understanding the effects of cattle grazing on rangeland fuel loads is particularly important.

Beef cattle account for the vast majority of rangeland cattle. However, the number of beef cows in California today is only about 57% of their peak numbers in the 1980s (Saitone 2018). This reduction is mirrored by declines in authorized grazing on public lands in the state over that time period (Oles et al. 2017; Saitone 2018). The number of grazed rangeland acres has been in decline as well, both on private (Cameron et al. 2014) and public lands (Forero 2002; Oles et al. 2017). This reduction influences rangeland fuel levels, as less fine fuel is removed through grazing.

Cattle grazing can reduce rangeland fuels in several ways. The most frequently studied and perhaps most important way is by removing fine fuels. This can affect fire behavior by reducing rates of spread, flame lengths and fire intensities. Despite widespread interest in this topic, there is only one published study of the impact of cattle grazing on fine fuels and fire behavior in California (Stechman 1983). This study looked at fire behavior in an annual grassland grazed by cattle; however, the level of residual dry matter (RDM) was much higher than is typical for grazed annual grassland in California. RDM is the amount of herbaceous plant matter from the previous season immediately prior to the first fall rains (Bartolome et al. 2006). Other studies from western U.S. rangelands in sagebrush steppe, mesquite savanna and cheatgrass-dominated grasslands have shown that cattle grazing can reduce fine fuel loads and, in turn, slow fire spread and flame length (Bruegger et al. 2016; Davies et al. 2010; Davies et al. 2015; Diamond 2009; Schmelzer et al. 2014). Several of these studies rely on fire behavior models to analyze the effects of fine fuel reduction on fire behavior (Bruegger et al. 2016; Diamond 2009).

Comparison of ungrazed grassland (inside exclosure) versus grazed grassland (outside exclosure). *Photo:* Royce Larsen.

Cattle grazing can also reduce rangeland fuels by causing long-term changes in species composition and vegetation structure. Perhaps the most important example of this in California is that cattle grazing can prevent or slow the encroachment of shrubs and trees into grassland. Much of coastal California has shown a trend of shrub encroachment on grassland (particularly by coyote brush, *Baccharis pilularis*) in the absence of grazing and fire disturbances (Ford and Hayes 2007). For instance, in the San Francisco Bay Area, limited grazing in the mid- to late 20th century has been linked to widespread shrub encroachment and loss of grassland (Keeley 2005; McBride and Heady 1968; Russell and McBride 2003). Coyote brush encroachment is also occurring on the southern California coast (Brennan et al. 2018). Shrub encroachment, even if by native species, presents a challenge for fire management because dense stands of shrubs increase fire hazard and fire intensity (Ford and Hayes 2007; Parker et al. 2016). Grazing is a key management technique to minimize these more severe wildfires in areas where retention of grasslands is an important goal.

The amount of herbaceous fuel on the ground during fire season in grazed California rangelands is largely a function of herbaceous growth in any given year, the number of livestock grazing per acre (grazing pressure), and vegetation biomass loss due to weathering (Frost et al. 2008; Larsen et al. 2021). Forage production is notoriously variable and unpredictable in California, both between years and across the landscape at a fine scale (Becchetti et al. 2016; Devine et al. 2019). The number of livestock grazing in the state is relatively stable by comparison.

The goals of this study are to inform planning, policy, and risk assessment at the state and regional scales and to clarify the benefit of strategic grazing to mitigate wildfire risk. To accomplish this, we describe the degree to which cattle remove fine fuels from rangelands in different areas of the state and use models to try to understand how this fine fuel removal affects fire behavior. We aim to help answer the following questions:

1. How much herbaceous fuel is removed by cattle from grazed rangelands in California, and how does this amount vary by region in the state?
2. What can fire behavior models tell us about how effective current levels of cattle grazing are at altering wildfire behavior?
3. How do spatial patterns of grazing and fuel reduction within regions inform our understanding of the impact of cattle grazing on fire behavior?

To answer the study questions, we first estimated rangeland fine fuel reduction by cattle in California. Next, we characterized year-to-year and spatial variability associated with fuel reduction. Finally, we applied fire models to predict how estimated regional fuel reduction would affect grassland fire behavior.

Calculating fuel reductions

We assumed that fine fuel reduction by cattle equals the amount of rangeland forage consumed by cattle in California. This is a conservative estimate of the total fuel reduction since it does not explicitly consider fine fuels removed through trampling (Nader et al. 2007), but see AUM in supplemental table 2 in the online supplemental appendix. Consumed rangeland forage is a function of the number of cattle grazing on rangelands (head), the class of cattle, and the time spent grazing on the rangeland (in months; equation 1). We used five datasets to determine the values in equation 1, including the 2017 USDA Agricultural Census, California Brand Inspection Data, County Crop Reports, GAP LANDFIRE vegetation classification and MODIS imagery (supplemental table 1). We also consulted with livestock and range advisors from the University of California Cooperative Extension (UCCE) to estimate irrigated pasture use and further refine the data (See "Animal Unit Months and Forage Removal" in the online supplemental appendix).

FIG. 1. Beef cattle grazing regions of California.

The census data provides an inventory of beef cows and "other cattle" in each county. "Other cattle" are all non-cow classes (including both beef and dairy cattle). We used the brand inspection data to estimate the proportion of "other cattle" that were beef cattle, and to estimate the proportion of these that belong to each non-cow class (supplemental tables 1 and 2).

In order to account for inter-county movement of cattle, we created beef production regions in California (fig. 1). These regions were selected to account for the majority of inter-county movements of cattle, and for similarities in forage production and livestock production practices for counties without pronounced patterns of inter-county cattle movement.

Regional rangeland acres were calculated by: (1) summing harvested rangeland acreage statistics from the county crop reports to estimate "Grazed Rangeland" acres, and (2) summing the rangeland acreage types per region using the GAP/LANDFIRE National Terrestrial Ecosystems (GAP) (USGS 2016) classification to estimate "Total Rangeland" acres.

We used the following equation to calculate the total pounds of forage removed on rangelands in each region by cattle (variables are described in supplemental table 2):

$$\textit{forage consumed} = \Sigma_{region\ k}\,(\Sigma_{county\ j}\,(\Sigma_{cattle\ class\ i}(\textit{head}_{ijk} \times \textit{months}_{ijk} \times \textit{AUE}_i - \textit{IP.adjust}_{ijk}) \times 1{,}000\ \textit{pounds/AUM}))$$

To estimate forage removed per rangeland acre, we divided the estimated forage consumed by rangeland acreage in each region. To account for differences in approaches to estimating rangeland acreage, we calculated this using two datasets: county crop reports and the GAP classification.

Forage production and RDM

RDM is the unused forage at the end of the grazing season (fall) (Bartolome et al. 2006), measured in pounds per acre or kilograms per hectare. The total amount of forage produced per acre on rangelands is generally measured in late spring at peak standing crop. It is an approximate measure of the amount of fine fuel produced per acre annually (excluding non-forage species), which is an important determinant of fuel load. RDM is not a perfect measure of fuel load because it excludes non-forage species and is only measured at the end of the fire season. Nevertheless, it gives an approximate value for residual fuel load. When compared to production measurements, RDM can be used to determine fine fuel removal rates by livestock in grazed rangelands.

We evaluated production data from 52 sites in the Central Coast, North Coast and Sacramento-Sierra-Cascade regions that was collected between 2000 and 2019, and RDM data from 105 sites collected between 1987 and 2019. We summarized these data to characterize variability in production between regions and at sub-regional scales, and to qualitatively assess heterogeneity of RDM and fuel reduction rates on grazed rangelands (supplemental table 4). We then compared these reduction rates to regional fuel reduction rates from the census-based fuel reduction estimates.

Modeling fire behavior

Custom fuel models were built using the BehavePlus 6 fire behavior model application to determine how variation in grassland fine fuel loads could affect flame length. Initial parameters were based on the low fuel load, dry-grass model GR2 (Scott and Burgan 2005), and the two grass models from the "original 13 fuel models" as described by Anderson (1982). However, several variables were altered to represent a range of fuel loads in different topographic positions and weather conditions (supplemental table 6). The pattern and scale of results from using the three different fuel models as the base for custom fuel models were similar (supplemental figs. 1–4). Therefore, our discussion is limited to the results of using the GR2 fuel model.

A summer model was built to represent fuel conditions after annual grasses had senesced and dried, and when fire conditions should be most extreme in a given year. For the summer models, we evaluated flame lengths when wind speeds were between 0 and 40 miles per hour (0–64.4 kilometers [km] per hour), and when fuel loads were between 100 and 2,000 pounds per acre (112–2,242 kilograms [kg] per hectare [ha]). Additionally, three separate dead fuel moisture scenarios (high at 13%, moderate at 6% and low at 2%) and two separate slope scenarios (high at 100% and low at 0%) were run. The high dead fuel moisture scenario was set to 13%, since our moisture of extinction (fuel moisture at which fuels are no longer ignitable) was set at 15% and is within the range of values that can be expected in California grasslands (Livingston and Varner 2016). While there is a dearth of literature on dead fuel moistures in California grasslands, the moderate dead fuel moisture scenario was set to 6%, because that was the lowest value measured by Livingston and Varner (2016) in late September. We set this as our moderate value, instead of our low value, because their measurements took place in Northern California, where we might expect higher dead fuel moistures due to a more mesic (moist) climate. Lastly, the low dead fuel moisture value was selected to represent very extreme fire conditions. The higher slope value of 100% slope was selected to represent a high slope scenario, but one that was still reasonable for firefighters to access.

A spring model that included more live fuel and a higher fuel moisture content was also evaluated (supplemental figs. 1 and 2). While the GR2 model is dynamic and automatically reapportions some of the live herbaceous fuel to a one-hour fuel load, we turned off the dynamic feature of our fuel models because we were manually setting the ratio of live to dead fuel as part of the spring and summer scenarios.

BehavePlus 6 defaults to setting a maximum effective windspeed, but studies have shown that this can underestimate flame lengths and rates of spread (Andrews et al. 2013). Therefore, we turned off this feature and did not impose a maximum effective windspeed in our model calculations. Additionally, BehavePlus 6 has the option for the windspeed to be calculated at the midflame height, 20 feet above the vegetation, or 10 meters above the vegetation. We set the input for wind speeds to be at midflame height. This is the average windspeed from the top of the fuel bed to the height of the flame in relation to the fuel.

Regional variations

Approximately 1.8 million beef cattle grazed rangelands in California in 2017. Although there was a slight dip in the number of beef cows in the state during the 2012–2015 drought, their number had rebounded to the decadal average by 2017 (CDFA 2010–2018), indicating that 2017 Census numbers are representative of the pre-drought cattle numbers.

Beef cows were by far the most abundant beef cattle class, with 677,000 on range in the state in 2017. This was followed by steers, heifers, "mixed" (an amalgamation of different classes that couldn't be separated using the brand inspection data), and bulls.

The number of months cattle spent on rangeland varied by county and by cattle class. Cows were estimated to spend an average of 10.7 months on rangeland (this accounts for cows that were removed from rangeland due to replacement). Steers and heifers were estimated to be on range an average of 7.6 and 7.7 months, respectively, and bulls and "mixed" cattle averaged 6.6 months on range. Time spent on range by each class of cattle varied substantially between counties and regions.

The cumulative fine fuel removal by these cattle varied by region from 85.0 million pounds (34.6 million kg) in the South Coast region to 5,444 million pounds (2,469 million kg) in the San Joaquin-Sierra region (fig. 2). In regions with higher levels of irrigated pasture use (San Joaquin-Sierra and Sacramento-Sierra-Cascade), estimates of fuel removal may be somewhat higher than actual removal rates if irrigated pasture use was higher

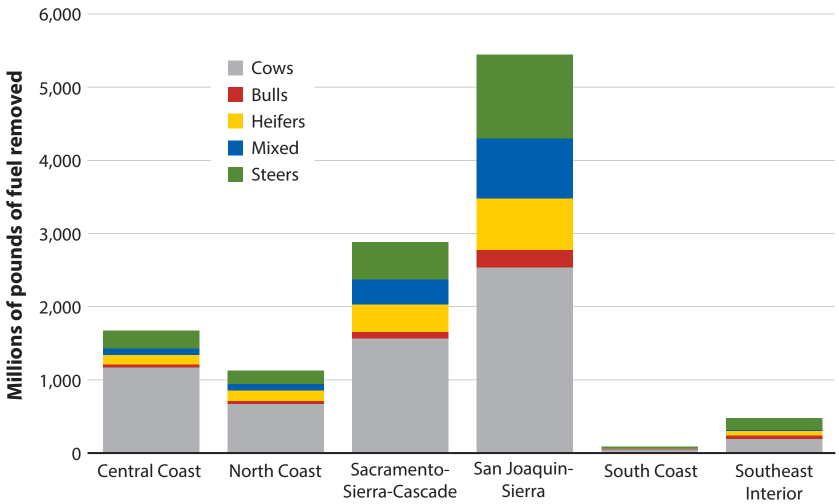


FIG. 2. Millions of pounds of rangeland fuel removed by cattle in each region.

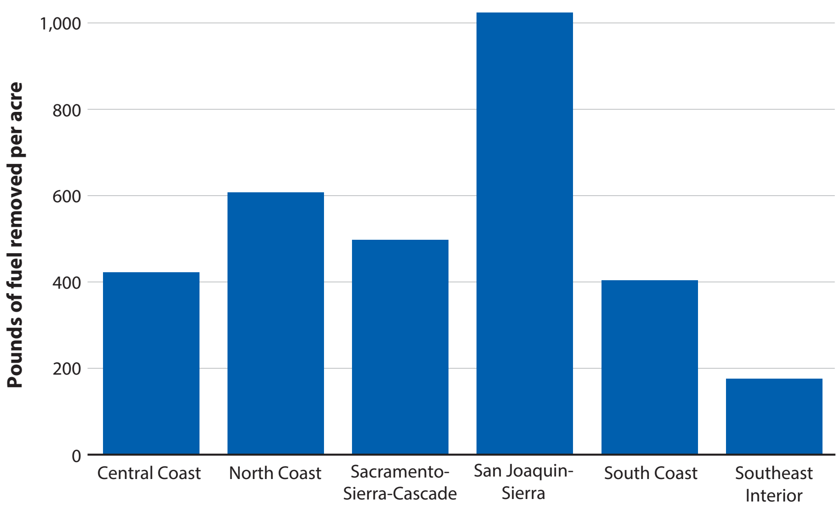


FIG. 3. Pounds per acre of fuel reduction on grazed rangelands in California regions.

in 2017 than the regional estimates used in our analysis. Across the state, the total fuel reduction by cattle in 2017 was 11.6 billion pounds (5.3 billion kg). Overall, this is probably a conservative estimate of fuels reduced on rangelands since it does not take into consideration fine fuels trampled by cattle and incorporated into mineral soil.

There were 19.4 million acres (7.9 million ha) of rangeland grazed by livestock in California according to county crop reports and county Agricultural Commissioners' offices. This is close to the 17 million acres (6.9 million ha) of private grazed rangeland previously reported in the state (CAL FIRE 2017), which is not surprising since many county crop reports do not include federal grazing allotments in their rangeland acreage estimates. On the other hand, our estimate of the total rangeland acreage based on the California GAP was 59.4 million acres (24 million ha). This estimate includes all public and privately owned rangeland, whether or not it is grazed.

The average amount of fuel removed across grazed rangelands in the state was 596 pounds per acre (668 kg/ha). This number varied from 174 pounds per acre (195 kg/ha) in the Southeast Interior region to 1,020 pounds per acre (1,143 kg/ha) in the San Joaquin-Sierra Region (table 1; fig. 3).

When calculated across all rangeland acres identified in the GAP analysis (not just grazed acres), average fuel reduction was only 195 pounds per acre (219 kg/ha). This lower number is largely due to the fact that there is rangeland that is not grazed in every region. The per-acre fuel reduction using the GAP acreage has similar regional trends to fuel reduction based on acreage from the county crop reports (table 1; fig. 4).

The regional values of grazing intensity are far below the amount of forage produced by region in most years. Valley grasslands in the interior of the state generally produce 2,000 pounds of forage per acre (2,242 kg/ha) or more in an average forage year (Bartolome 1987; Becchetti et al. 2016). Central and northern coast

TABLE 1. Acreage and average fuel reduction rates on grazed and total rangelands by region

Region	Grazed rangeland acreage (from crop reports)	All rangeland acreage (from GAP)	Fuels removed – grazed rangelands (pounds/acre)	Fuels removed – all rangelands (pounds/acre)
Central Coast	3,983,153 (1,611,925 ha)	7,242,014 (2,930,739 ha)	419 (470 kg/ha)	230 (258 kg/ha)
North Coast	1,857,912 (751,870 ha)	2,504,836 (1,013,671 ha)	419 (470 kg/ha)	450 (504 kg/ha)
Sacramento-Sierra-Cascade	5,827,095 (2,358,142 ha)	11,703,394 (4,736,196 ha)	495 (555 kg/ha)	246 (276 kg/ha)
San Joaquin-Sierra	5,336,824 (2,159,736 ha)	9,265,683 (3,749,689 ha)	1,020 (1143 kg/ha)	588 (659 kg/ha)
South Coast	211,560 (85,615 ha)	3,659,608 (1,480,991 ha)	401 (449 kg/ha)	23 (26 kg/ha)
Southeast Interior	2,232,720 (903,550 ha)	25,031,549 (10,129,908 ha)	174 (195 kg/ha)	16 (18 kg/ha)
Total	19,449,264 (7,870,838 ha)	59,407,085 (24,041,194 ha)	596 (average) (668 kg/ha)	195 (average) (219 kg/ha)

range grassland sites produce more than 3,000 pounds of forage per acre (3,363 kg/ha) (Becchetti et al. 2016; Larsen et al. 2020). Coastal prairie sites can be highly productive, producing more than 4,500 pounds per acre (5,044 kg/ha) on average in the Central Coast (Larsen et al. 2020). In the highest production years, forage production can be double the average in any given region, and in the lowest production years it can be less than 25% of average production (Larsen et al. 2020). The relatively low grazing intensity reflects the generally conservative stocking strategies used by many ranchers across the state to hedge against the unpredictable and highly variable annual forage production (Macon et al. 2016).

It's important to keep in mind that grazed acres and forage removal rates in this paper are not "hard numbers," but rather are estimates to inform large-scale patterns of fuel removal by cattle. These estimates are based on the best available data, but these data do not describe the intricate (and dynamic) details of cattle grazing across the state. These numbers should be interpreted in the context of understanding regional fuel reduction, not as predictive of grazing practices at subregional scales. There is a need for more consistent and accurate reporting of cattle numbers and grazed acres across the state.

Based on several datasets, forage production and RDM were highly variable within and between regions of the state. Average RDM in each region was significantly less than production, but the amount of fuel reduced was highly variable (table 2).

Collectively, these data show that reductions of fuels measured on ranches can differ significantly from region-wide averages seen in the Census analysis. The Census gives an indication of the county in which grazing occurs, but it does not tell us where those animals graze within the county. The RDM data also show that spatial differences in forage production and grazing practices can lead to differences in the amount of fine fuels and the level of fuel reduction by cattle. This is consistent with other research showing that annual forage production is highly variable across the state, varying at small and large scales in relation to soil characteristics, microclimate, position on the landscape, and tree canopy cover (Becchetti et al. 2016; Devine et al. 2019; Frost et al. 1991).

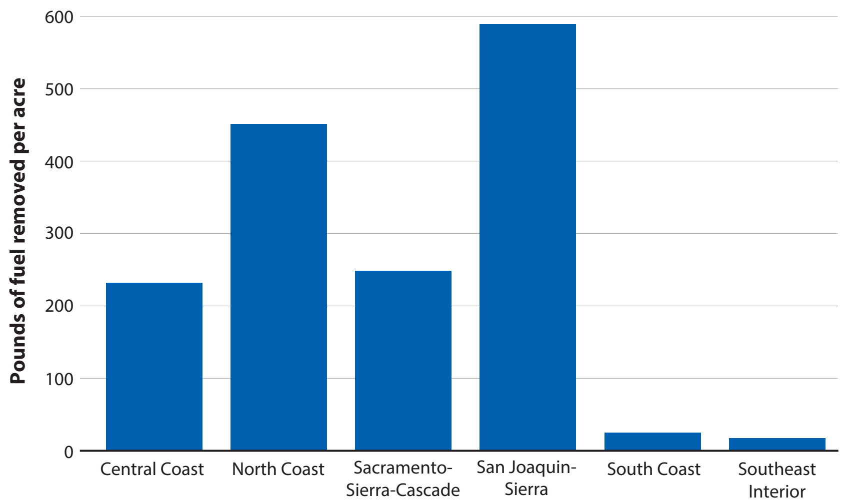


FIG. 4. Pounds per acre of fuel reduction on all rangelands in California regions.

Lower flame lengths

Keeping flame lengths below eight feet (2.4 meters [m]) is seen as a critical threshold that allows fire fighters to use direct measures (such as heavy equipment) on the ground to fight fires. Below four feet (1.2 m), fires can be fought using hand tools (Andrews and Rothermel 1982). However, these thresholds are somewhat fuzzy and dependent on other aspects of the fire, i.e.,

TABLE 2. Forage production and residual dry matter (RDM) from coastal prairie, coast range grassland, and valley grassland sites in Central and Northern California

Region	Data source	Average production (pounds/acre)	Production minus summer decomposition (75% of total)*	Average RDM (pounds/acre)	Average fuel reduction (pounds/acre)
Central Coast (Coastal)	Larsen et al. 2020	4,978 (5,580 kg/ha)	3,734 (4,185 kg/ha)	1,815 (2,034 kg/ha)	1,919 (2,151 kg/ha)
Northern California (Coastal)	Bartolome et al. 2015 and Point Reyes unpublished data 2020	7,053† (7,905 kg/ha)	5,290 (5,929 kg/ha)	2,147 (2,406 kg/ha)	3,143 (3,523 kg/ha)
Central Coast (Coast Range)	Larsen et al. 2020	3,371 (3,778 kg/ha)	2,528 (2,834 kg/ha)	2,055 (2,303 kg/ha)	473 (530 kg/ha)
Central Coast (Coast Range)	NRCS unpublished data 2010	3,055 (3,424 kg/ha)	2,138 (2,396 kg/ha)	1,775 (1,990 kg/ha)	363 (407 kg/ha)
Central Coast (Interior)	Larsen et al. 2020	1,961 (2,198 kg/ha)	1,471 (1,649 kg/ha)	1,053 (1,180 kg/ha)	418 (469 kg/ha)
Sacramento-Sierra-Cascade (Interior)	UC ANR unpublished data	3,096 (3,470 kg/ha)	2,322 (2,603 kg/ha)	800‡ (897 kg/ha)	1,522 (1,706 kg/ha)

* Based on Frost et al. 2005.
† Production values from only two years of data.
‡ RDM values estimated not measured.

spread and fire intensity (Andrews et al. 2011). Based on our fire behavior models, on flat ground in dry summer conditions (when dead fuel moisture is 6%), fine fuel loads below 1,225 pounds per acre (1,373 kg/ha; fig. 5) are predicted to keep flame lengths below eight feet at wind speeds up to 15 miles per hour (24 km per hour). At higher dead fuel moisture levels and lower wind speeds, flame lengths may be kept below eight feet at higher fuel loads. However, in extreme fire weather with very low dead fuel moisture (2%) and wind speeds up to 40 miles per hour (64.4 km per hour), fine fuel loads may need to be reduced below 214 pounds per acre (240 kg/ha) (fig. 5) to keep flame lengths under eight feet. In high slope areas during dry conditions (6% dead fuel moisture) with windspeeds of 15 miles per hour, fine fuel loads would need to be kept below 1,000 pounds per acre (1,121 kg/ha) to keep flame lengths below eight feet. In very dry conditions (2% dead fuel moisture), at wind speeds of 40 miles per hour, fuel loads would need to be reduced below 205 pounds per acre (230 kg/ha) to keep flame lengths below eight feet. While these models are useful for interpreting potential impacts of estimated fuel reduction levels, the results still need to be experimentally validated in California before they are used for policy and planning purposes. Also, these models do not evaluate ignition potential, level of shrub encroachment, and areas with elevated ignition risk, which may have different fuel load thresholds. There is always a level of uncertainty associated with fire behavior modeling. Depending on the aptness of the fuel models, BehavePlus 6 results can be off by a factor of two or more (Sparks et al. 2007).

Understanding the effect of cattle grazing on fire behavior is complicated by the pronounced spatial and temporal variability in forage production, fuel reduction, shrub encroachment and RDM at scales smaller than the region or county. In their measurements at 43 different ranches spanning a rainfall gradient in Central California, Larsen et al. (2020) found RDM values ranging from 75 to 6,258 pounds per acre (84 to 7,014 kg/ha) from 2000 to 2019. Forty percent of grazing fields had RDM values at or below 1,225 pounds per acre (1,373 kg/ha), while only 4% were below 214 pounds per acre (240 kg/ha). This shows that many areas of these grazed rangelands had good fuel conditions for non-extreme fire weather, but few locations had fuel levels low enough to keep flame lengths below eight feet in extreme fire weather. No grazing fields had RDM below these thresholds consistently across all monitoring years.

Strategic grazing

The inherent heterogeneity of grazing intensity and fuel reduction may in fact be its greatest asset in reducing wildfire hazard and risk. Selective grazing by livestock can create patchiness of fuels, reducing continuity of fuels and reducing rate of fire spread and total burned area (Bunting et al. 1987; Kerby et al. 2007;

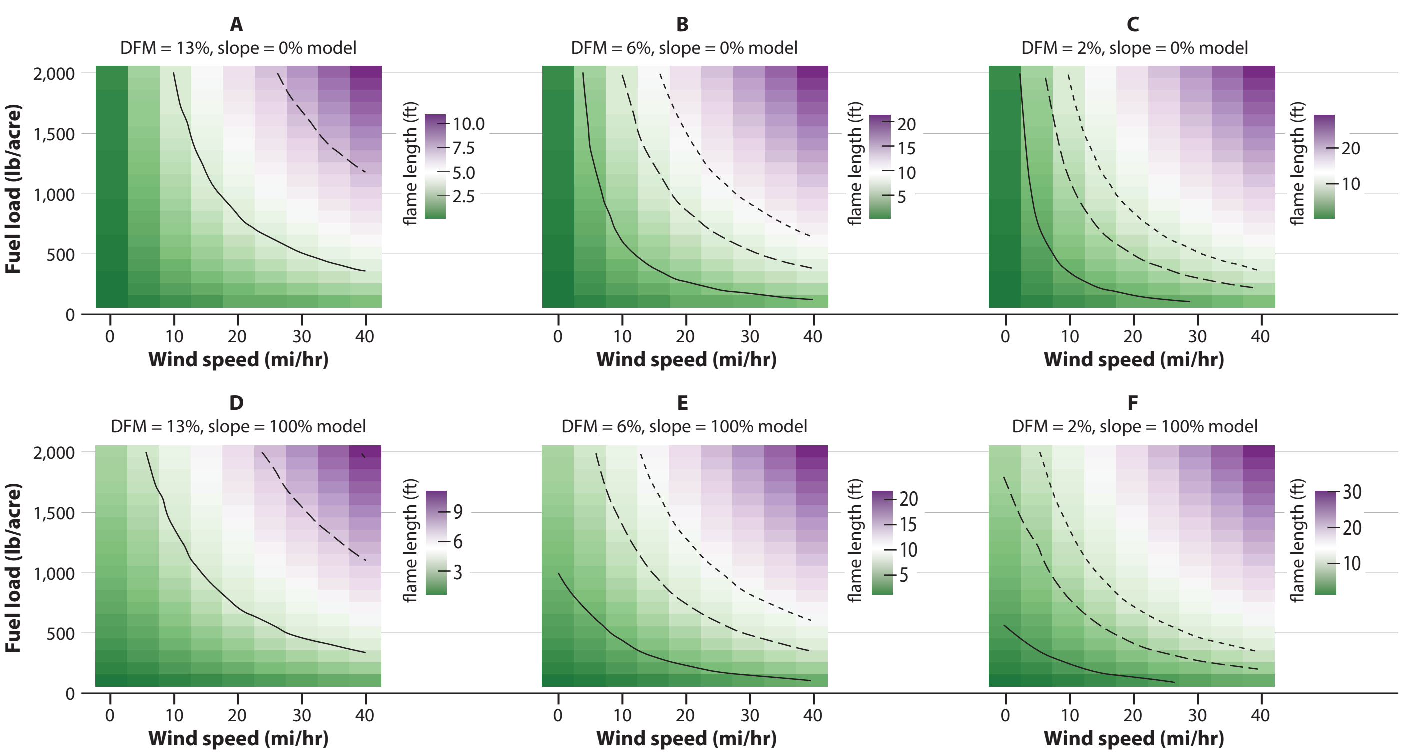


FIG. 5. Results from fire behavior modeling under summer conditions. Conditions were run under three dead fuel moisture scenarios of 13% (A, D), 6% (B, E) and 2% (C, F), and two slope scenarios of 0% (A, B and C) and 100% (D, E and F). Contour lines show when threshold flame lengths of 4 feet (solid line), 8 feet (long-dashed line) and 11 feet (short-dashed line) are surpassed.

This cow-calf operation on the Central Coast has cattle grazing on the ranch year-round, helping to reduce the potential for catastrophic wildfire. *Photo:* Devii Rao.

Launchbaugh 2016; Taylor 2006). At the ranch scale, RDM data from the Central Coast shows that, even in a region with relatively low grazing intensity, fuel reduction of several thousand pounds per acre can be achieved in select locations (Larsen et al. 2020).

Given that grazing intensity on California rangelands is generally conservative relative to the amount of forage produced in most years (as evidenced by the generally low fuel reduction for most regions in the Census analysis), strategic implementation of grazing should be employed to maximize the benefit of livestock grazing for fuels reduction. A strategic grazing program would target grazing on certain areas of the landscape. It should consider maintaining fuel breaks, controlling shrub encroachment, employing grazing near the wildland-urban interface, proximity to urban centers, annual weather patterns (i.e., grazing in advance of Santa Ana or Diablo winds), potential sources of ignition, and the realities of grazing operations (including animal distribution, nutrition, site accessibility, and the need to bank forage for the fall). To be successful, grazing strategies must be logistically feasible and financially sustainable for the grazing operator.

A strategic approach to fuels reduction is especially important given that California rangelands are managed for multiple resource objectives. Reducing fuels on all grazed rangelands to 1,225 pounds per acre (1,373 kg/ha) or less will not be compatible with some of these objectives in some areas. RDM recommendations are based on the type of grassland (dry annual grassland, annual grasslands/hardwood rangeland, or coastal prairie), terrain slope, and percent cover of woody vegetation (Bartolome et al. 2006). RDM standards vary from 300 pounds per acre (336 kg/ha) on some dry, flat inland sites to 2,100 pounds per acre (2,354 kg/ha) on steep, coastal prairie sites (Bartolome et al. 2006). Maintaining adequate RDM is expected to minimize soil erosion, improve forage production, and influence plant species composition at some sites — but many areas have RDM standards above the preliminary fuel load thresholds reported here. In particular, steeper areas have higher minimum RDM recommendations — but these areas would need even lower fuel loads to keep flame lengths below eight feet. Testing these fuel load thresholds on the ground and having discussions between fire modelers and rangeland specialists will be critical to making appropriate recommendations about grazing levels to achieve both fire safety and natural resource objectives. Furthermore, RDM is measured immediately prior to the first germinating rains (September or October) and fuel reductions will need to be achieved earlier in the year if they are meant to apply to the bulk of the fire season. Fuel reduction also must ensure that adequate forage is left to support continued livestock grazing during the fall and winter months.

There are several potential synergies between reducing residual biomass for fire safety and conservation objectives. Excessive residual biomass and height have been found to negatively affect many sensitive or threatened wildlife species (Ford et al. 2013; Gennet et al. 2017; Germano et al. 2011; Riensche 2008), cause problems for weed management (Becchetti et al. 2016), and negatively affect some native plant species (Bartolome et al. 2014; Beck et al. 2015). Where possible, maximum biomass standards for fuel reduction should be strategically implemented to simultaneously promote these and other conservation goals.

Cattle grazing is not the only management tool that can be used to reduce residual biomass. Unlike wildfires, prescribed fires are well planned, and are implemented to achieve one or more specific objectives. Prescribed fires burn thatch, increasing seed access to the soil surface, and creating more suitable light conditions and ground temperatures for grassland forbs (Sugihara et al. 2006). This allows higher levels of seed production and flowering in forbs after late spring fires. Prescribed fire can be used alone, or in conjunction with grazing, to improve habitat for some native plants and sensitive or threatened wildlife species. In the early 1950s, ranchers were permitted to burn a substantial amount of land in California, up to more than 200,000 acres in one year (Biswell 1999). Since that time, prescribed burn acreage has been in steep decline. However, due to recent catastrophic wildfires, there is renewed interest in prescribed burning. Though grazing is substantially more widespread than prescribed burning today, thanks to new legislation (SB 901 and SB 1260) and development of prescribed burn associations across the state, prescribed burning is becoming a viable option again.

Grazing can reduce fuel

Cattle grazing plays an important role in reducing fuels on California rangelands. Without grazing, we would have hundreds or possibly thousands of additional pounds per acre of fuel on rangelands, potentially leading to larger and more devastating fires. Cattle grazing, of course, can't eliminate wildfires completely. But it can make a big impact. Cattle don't consume forage uniformly on rangelands. Instead, they eat in more of a patchwork pattern. Thus, while cattle grazing does not reduce fuels enough to avoid hazardous 4- or 8-foot wildfire flame lengths on all grazed rangelands, many areas will be grazed sufficiently to significantly alter fire behavior (especially in non-extreme fire weather).

To effectively reduce wildfire hazards, rangeland managers and planners must strategically coordinate fuel management practices, such as cattle grazing along with other natural resource objectives and management practices, including prescribed fire. This will require the development of maximum residual biomass standards that can be used to assess fuel loads at critical times and locations during the fire season. To help develop these standards, we need to experimentally validate fire behavioral models in herbaceous rangelands in California.

Widespread wildfires are predicted to increase over time in California due to ongoing climate change. This new reality requires that we take advantage of all the tools available to protect public safety while also meeting broader rangeland management objectives. All of this is occurring against the backdrop of the decline of the number of beef cows grazing in California, including on public lands, over the past several decades (Oles et al. 2017; Saitone 2018). It is not feasible to graze all rangelands to ideal fuel levels, nor is it compatible with management goals across the state. However, there are opportunities to improve fire safety in California by grazing rangelands that are not currently being grazed — or even by increasing grazing intensity on some very lightly grazed areas. Strategic implementation of cattle grazing, including potentially fee-for-service agreements on key private and public lands, can meet multiple natural resource objectives while also lowering fire hazards by reducing fine fuels, reducing fuel continuity and slowing or even stopping shrub encroachment onto grasslands. CA

F. Ratcliff is Rangeland Conservation Scientist, LD Ford, Consultants in Rangeland Conservation Science; D. Rao is UC Cooperative Extension (UCCE) Livestock and Natural Resources Advisor, San Benito, Monterey and Santa Cruz counties; S. Barry is UCCE Livestock and Natural Resources Advisor, Santa Clara, San Mateo, Alameda and Contra Costa counties; S. Dewees is Graduate Student, Ecology, Evolution, Marine Biology, UC Santa Barbara; L. Macaulay is Wildlife Management Specialist, University of Maryland Extension; R. Larsen is UCCE Area Natural Resource/Watershed Advisor, San Luis Obispo County; M.W.K. Shapero is UCCE Livestock and Range Advisor, Ventura and Santa Barbara counties; R. Peterson is M.S. Student, Graduate Group in Ecology, UC Davis; M. Moritz is UCCE Wildfire Specialist, Bren School, UC Santa Barbara; L. Forero is UCCE Livestock and Natural Resources Advisor, Shasta and Trinity counties.

This project was funded by the California Cattle Council. The authors would like to thank Dan Macon, Tracy Schohr, John Harper, Scott Oneto, Laura Snell, Morgan Doran, Julie Finzel, Josh Davy, Rebecca Ozeran, David Lile, Jeffery Stackhouse, Theresa Becchetti, Brooke Latack, Rob Wilson, Fadzayi Mashiri, Callie Peek, Chris McDonald and Stephanie Larson for providing irrigated pasture and regional cattle production information.

References

Anderson HE. 1982. Aids to Determining Fuel Models for Estimating Fire Behavior. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Andrews PL, Cruz MG, Rothermel RC. 2013. Examination of the wind speed limit function in the Rothermel surface fire spread model. Int J Wildland Fire 22:959. https://doi.org/10.1071/WF12122

Andrews PL, Rothermel RC. 1982. Charts for interpreting wildland fire behavior characteristics. General Technical Report INT-131. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Andrews, PL, Heinsch FA, Schelvan L. 2011. How to Generate and Interpret Fire Characteristics Charts for Surface and Crown Fire Behavior. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. General Technical Report RMRS-GTR-253.

Bartolome JW. 1987. California annual grassland and oak savannah. Rangelands 9:122–5.

Bartolome JW, Frost WE, McDougald NK, Connor M. 2006. California Guidelines for Residual Dry Matter (RDM) Management on Coastal and Foothill Annual Grasslands. Publication 8092. University of California, Division of Agriculture and Natural Resources, Rangeland Monitoring Series.

Bartolome JW, Allen-Diaz BH, Barry S, et al. 2014. Grazing for biodiversity in Californian Mediterranean grasslands. Rangelands 36:36–43. https://doi.org/10.2111/Rangelands-D-14-00024.1

Bartolome, J, Hammond M, Hopkinson P, Ratcliff F. 2015. 1987–2014 Residual Dry Matter Analysis Report and Updated Rangeland Monitoring Guidelines for Livestock Grazed Grasslands within Point Reyes National Seashore and Golden Gate National Recreation Area. Produced by the UC Berkeley Rangeland Ecology Lab for Point Reyes National Seashore.

Becchetti T, George M, McDougald N, et al. 2016. Rangeland Management Series: Annual Range Forage Production. Publication 8018. University of California, Agriculture and Natural Resources. 12 p. https://doi.org/10.3733/ucanr.8018

Beck JJ, Hernández DL, Pasari JR, et al. 2015. Grazing maintains native plant diversity and promotes community stability in an annual grassland. Ecol Appl 25:1259–70. https://doi.org/10.1890/14-1093.1

Biswell H. 1999. *Prescribed Burning in California Wildlands Vegetation Management.* Berkeley and Los Angeles: University of California Press.

Brennan S, Laris PS, Rodrigue CM. 2018. Coyote brush as facilitator of native California plant recovery in the Santa Monica Mountains. Madroño 65:47–59. https://doi.org/10.3120/0024-9637-65.1.47

Bruegger RA, Varelas LA, Howery LD, et al. 2016. Targeted grazing in Southern Arizona: Using cattle to reduce fine fuel loads. Rangeland Ecol Manag 69:43–51. https://doi.org/10.1016/j.rama.2015.10.011

Bunting SC, Kilgore, BM, Bushey CL. 1987. Guidelines for Prescribed Burning Sagebrush-grass Rangelands in the Northern Great Basin. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

[CAL FIRE] California Department of Forestry and Fire Protection. 2017. Fire and Resource Assessment Program (FRAP).

[CDFA] California Department of Food and Agriculture. 2010–2018. *California Agricultural Statistics Review*. Sacramento, CA: California Department of Food and Agriculture.

Cameron DR, Marty J, Holland RF. 2014. Whither the rangeland? Protection and conversion in California's rangeland ecosystems. PLOS ONE 9:e103468. https://doi.org/10.1371/journal.pone.0103468

Davies KW, Bates JD, Svejcar TJ, et al. 2010. Effects of long-term livestock grazing on fuel characteristics in rangelands: An example from the Sagebrush steppe. Rangeland Ecol Manag 63:662–9. https://doi.org/10.2111/REM-D-10-00006.1

Davies KW, Boyd CS, Bates JD, et al. 2015. Dormant season grazing may decrease wildfire probability by increasing fuel moisture and reducing fuel amount and continuity. Int J Wildland Fire 24:849. https://doi.org/10.1071/WF14209

Devine SM, O'Geen AT, Larsen RE, et al. 2019. Microclimate–forage growth linkages across two strongly contrasting precipitation years in a Mediterranean catchment. Ecohydrology 12. https://doi.org/10.1002/eco.2156

Diamond JM, Call CA, Devoe N. 2009. Effects of targeted cattle grazing on fire behavior of cheatgrass-dominated rangeland in the northern Great Basin, USA. Int J Wildland Fire 18:944. https://doi.org/10.1071/WF08075

[EBMUD] East Bay Municipal Utilities District. 2000. East Bay Watershed Fire Management Plan. Oakland, CA: East Bay Municipal Utilities District.

[EBRPD] East Bay Regional Parks District. 2013. East Bay Regional Parks District Master Plan 2013. Oakland, CA: East Bay Regional Parks District.

Ford LD, Hayes GF. 2007. Northern coastal scrub and coastal prairie. In *Terrestrial Vegetation of California, 3rd Edition*. Barbour M (ed.). University of California Press. p 180–207.

Ford LD, Van Hoorn P, Rao DR, et al. 2013. Managing Rangelands to Benefit California Red-Legged Frogs and California Tiger Salamanders. Livermore, CA: Alameda County Resource Conservation District.

Forero LC. 2002. Grass, Graziers, and Tenure: A Case Study on the Shasta-Trinity National Forest. Dissertation. Berkeley, CA: University of California, Berkeley, Department of Environmental Science Policy and Management (ESPM).

Frost WE, Bartolome JW, Churches KR. 2005. Disappearance of residual dry matter (RDM) on annual rangelands in the absence of grazing. XX International Grassland Conference. Dublin, Ireland.

Frost WE, McDougald NK, Demment MW. 1991. *Blue Oak Canopy Effect on Seasonal Forage Production and Quality*. Davis, CA: USDA Forest Service.

Frost WE, McDougald NK, Larsen R, et al. 2008. Disappearance of residual dry matter on coastal and Sierran annual rangeland of California. In Society for Range Management: Building Bridges Grasslands to Rangelands. Louisville, KY. Abstract 2435.

Gennet S, Spotswood E, Hammond M, Bartolome JW. 2017. Livestock grazing supports native plants and songbirds in a California annual grassland. PLOS ONE 12:e0176367. https://doi.org/10.1371/journal.pone.0176367

George M, McDougald N. 2010. Bitter Creek National Wildlife Refuge Independent Rangeland Review. Bitter Creek National Wildlife Refuge.

Germano, DJ, Rathburn GB, Saslaw LR. 2011. Effects of grazing and invasive grasses on desert vertebrates in California. J Wildlife Manage 76(4):670–82. https://doi.org/10.1002/jwmg.316

Green LR, Newell LA. 1982. Using goats to control brush regrowth on fuelbreaks. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station.

Keeley JE. 2005. Fire history of the San Francisco East Bay region and implications for landscape patterns. Int J Wildland Fire 14:285. https://doi.org/10.1071/WF05003

Kerby JD, Fuhlendorf SD, Engle DM. 2007. Landscape heterogeneity and fire behavior: Scale-dependent feedback between fire and grazing processes. Landscape Ecol 22:507–16. https://doi.org/10.1007/s10980-006-9039-5

Launchbaugh KL. 2016. Targeted grazing to manage wildland fuels and alter fire behaviour. In Proceedings: 10th International Rangeland Conference. Iwaasa A, Lardner HA, Schellenberg M, et al. (eds.). Saskatoon, SK. p 674–5.

Launchbaugh KL, Walker J. 2006. Targeted grazing — A new paradigm for livestock management. In *Targeted Grazing: A Natural Approach to Vegetation Management and Landscape Enhancement*. Launchbaugh KL, Walker J., Daines RL (eds.). Centennial, CO: American Sheep Industry Association.

Larsen R, Shapero M, Horney M, et al. 2020. Forage Production Report, California Central Coast, 2001–2019. University of California Agriculture and Natural Resources. http://cesanluisobispo.ucanr.edu/Custom_Program355/Forage_Production_Report/ (accessed August 2020).

Larsen RE, Shapero MWK, Striby K, et al. 2021. Forage quantity and quality dynamics due to weathering over the dry season on California annual rangelands. Rangeland Ecol Manag 76:150–6. https://doi.org/10.1016/j.rama.2021.02.010

Livingston AC, Varner JM. 2016. Fuel moisture differences in a mixed native and non-native grassland: Implications for fire regimes. Fire Ecol 12:73–87. https://doi.org/10.4996/fireecology.1201073

Macon DK, Barry S, Becchetti T, et al. 2016. Coping with drought on California rangelands. Rangelands 38(4):222–8. https://doi.org/10.1016/j.rala.2016.06.005

McBride J, Heady HF. 1968. Invasion of grassland by *Baccharis pilularis* DC. J Range Manage 21:106. https://doi.org/10.2307/3896366

Minnich RA. 1982. *Grazing, Fire, and the Management of Vegetation on Santa Catalina Island, California*. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture.

Nader G, Henkin Z, Smith E, et al. 2007. Planned herbivory in the management of wildfire fuels. Rangelands 29(5):18–24. https://ucanr.edu/sites/UCCE_LR/files/203022.pdf

Narvaez N. 2007. Prescribed herbivory to reduce fuel load in California chaparral. Doctoral dissertation, Department of Ecology, UC Davis.

[NRCS] National Resources Conservation Service. 2010. Unpublished data from the project Central Coast Rangeland Coalition Indicators of Sustainable Rangeland Stewardship.

Oles KM, Weixelman DA, Lile DF, et al. 2017. Riparian meadow response to modern conservation grazing management. Environ Manage 60:383–95. https://doi.org/10.1007/s00267-017-0897-1

Parker VT, Pratt RB, Keeley JE. 2016. Chaparral. In *Ecosystems of California—A Source Book*. Mooney H, Zavaleta E (eds.). Berkeley, CA: University of California Press. p 479–508.

[Rancho Mission Viejo] County of Orange, California Department of Fish and Game and U.S. Fish and Wildlife Service. 2006. *Rancho Mission Viejo Grazing Management Plan*. Appendix G in Southern Orange County HCP/MSAA/NCCP. p G1–G42.

Riensche DL. 2008. Effect of cattle grazing on lizard diversity in California grasslands. T W Sec Wil 44:4–10.

Russell WH, McBride JR. 2003. Landscape scale vegetation-type conversion and fire hazard in the San Francisco Bay Area open spaces. Landscape Urban Plan 64:201–8. https://doi.org/10.1016/S0169-2046(02)00233-5

Saitone TL. 2018. Livestock and rangeland in California. In *California Agriculture: Dimensions and Issues*. Martin PL, Goodhue RE, Wright BD (eds.). UC Berkeley: Giannini Foundation of Agricultural Economics. p 18.

Santa Clara County Parks. 2018. Santa Clara County Parks 2018 Strategic Plan.

Schmelzer L, Perryman B, Bruce B, et al. 2014. Case study: Reducing cheatgrass (*Bromus tectorum* L.) fuel loads using fall cattle grazing. Prof Anim Sci 30:270–8. https://doi.org/10.15232/S1080-7446(15)30112-1

Scott JH, Burgan RE. 2005. *Standard Fire Behavior Fuel Models: A Comprehensive Set for Use with Rothermel's Surface Fire Spread Model*. U.S. Department of Agriculture, Forest Service Rocky Mountain Research Station. General Technical Report RMRS-GTR-153.

Sparks JC, Masters RE, Engle DM, et al. 2007. Comparison of BEHAVE: Fire behavior prediction and fuel modeling system predictions with observed fire behavior varying by season and frequency. In Proceedings of the 23rd Tall Timbers Fire Ecology Conference: Fire in Grassland and Shrubland Ecosystems. Tallahassee, FL: Tall Timbers Research Station. p 170–80.

Stechman JV. 1983. Fire hazard reduction practices for annual-type grassland. Rangelands 5:56–8.

Sugihara N, Wagtendonk J, Fites-Kaufman K. 2006. Fire as an ecological process. In *Fire in California's Ecosystems*. Sugihara N, Wagtendonk J, Fites-Kaufman K, et al. (eds.). University of California Press. p 58–75.

Taylor CA. 2006. Targeted grazing to manage fire risk. In *Targeted Grazing: A Natural Approach to Vegetation Management and Landscape Enhancement*. Launchbaugh KL (ed.). Denver, CO: American Sheep Industry Association. p 107–14.

Tsiouvaras CN, Havlik NA. Bartolome JW. 1989. Effects of goats on understory vegetation and fire hazard. Forest Sci 35:1125–31.

[USDA] United States Department of Agriculture National Agriculture Statistics Service. 2017. 2017 Census of Agriculture. Washington, DC: USDA National Agriculture Statistics Service.

[USFS] United States Department of Agriculture Forest Service. 2022. Fire terminology. www.fs.fed.us/nwacfire/home/terminology.html (accessed June 14, 2022).

[USGS] United States Geological Survey Gap Analysis Project. 2016. GAP/LANDFIRE National Terrestrial Ecosystems 2011. https://doi.org/10.5066/F7Z-S2TM0

Van Soest PJ. 1994. *Nutritional Ecology of the Ruminant*. New York, NY: Comstock Publishing Associates, Cornell University Press.

RESEARCH ARTICLE

Winter flooding recharges groundwater in almond orchards with limited effects on root dynamics and yield

Almond orchards on soils with moderately high SAGBI or better can likely be used for winter water recharge with minimal negative effects and potentially some horticultural benefits.

by Xiaochi Ma, Helen Dahlke, Roger Duncan, David Doll, Paul Martinez, Bruce Lampinen and Astrid Volder

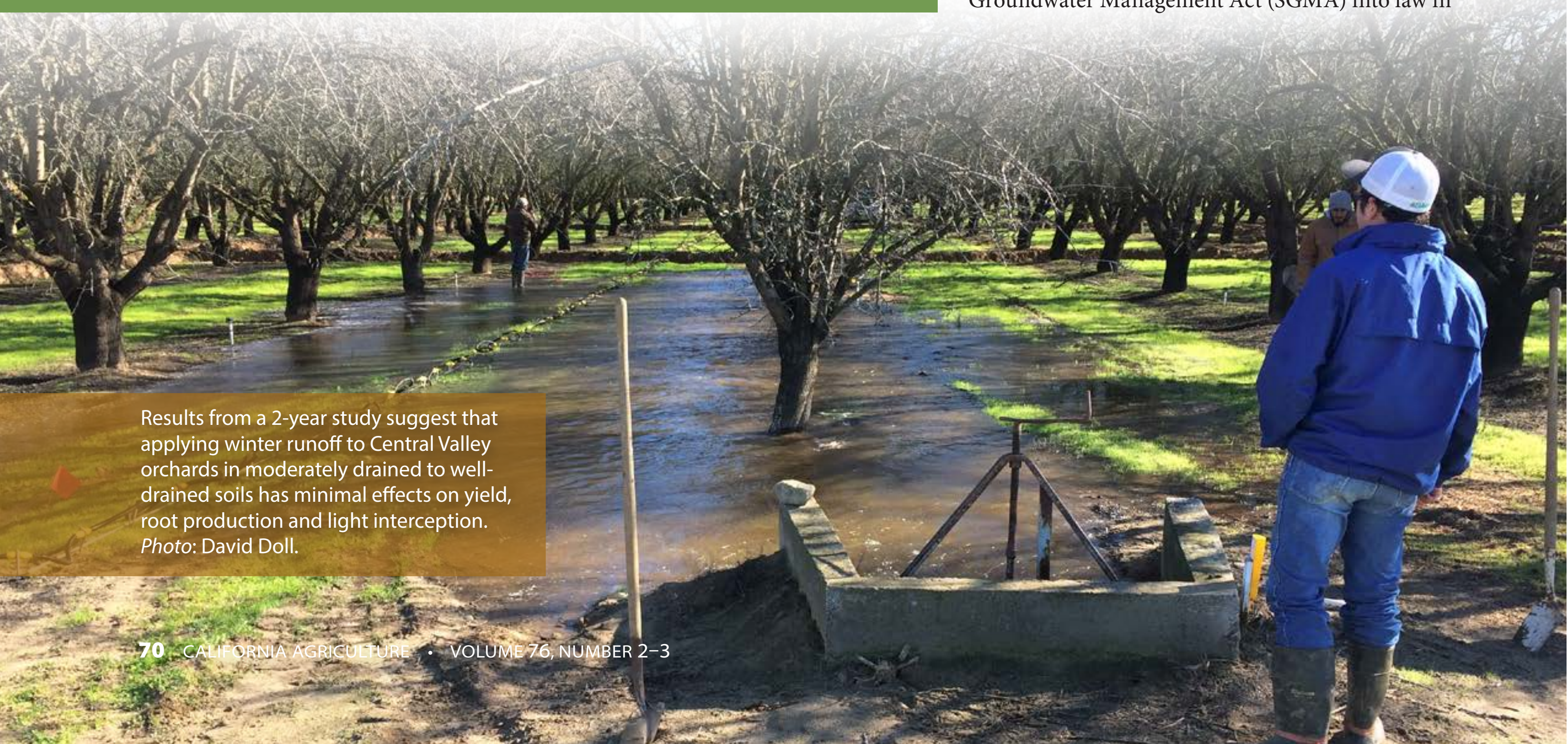
Online: https://doi.org/10.3733/ca.2022a0008

Abstract

California signed the Sustainable Groundwater Management Act (SGMA) into law in 2014. SGMA requires groundwater-dependent regions to halt overdraft and develop plans to reach an annual balance of pumping and recharge. Groundwater aquifers can be recharged by flooding agricultural fields when fallow, but this has not been an option for perennial crops such as fruit and nut trees. While flooding these crops might be possible during the dormant season, it is not known what impact flooding might have on tree-root systems, health and yield. We followed root production, tree water status and yield in two almond orchards in Northern California for 2 years to test the impact of applying captured winter water runoff for groundwater recharge purposes on tree performance. Results showed that more than 90% of the water applied to sandy soil and 80% of the water applied to loamy soil percolated past the root zones, with no measured adverse effects on tree water status, canopy development or yield. Groundwater recharge did not negatively affect new root production and tended to extend root lifespan. Based upon these data, applying additional water in late December and January is not likely to have negative impacts on almond orchards in moderately drained to well-drained soils.

Almond (*Prunus* spp.) is one of the top producing commodities in California; in 2019, almonds provided producers with cash receipts of $6.09 billion (CDFA 2020). From 2010 to 2019, almond acreage in the state increased by 79%; acreage of trees 4 years and older — called bearing acres — increased by 53%. During the same period, total California almond production increased by 55%, with an approximate value increase of $3.2 billion (CDFA 2020).

The expansion of almond orchards has increased irrigation demand in areas that rely heavily on groundwater reserves. In spite of some high water years (2017, 2019), the 10-year trend (2010–2020) shows that 28.4% of monitored wells had a water level decrease of 5 to 25 feet and 9.6% of monitored wells decreased by more than 25 feet. Over that same period, 14.8% of wells showed an increase in groundwater level (CADWR 2019). Groundwater decreases are particularly pronounced in the Tulare Lake, San Joaquin River and Sacramento River hydrologic regions (the whole Central Valley). In an effort to reduce groundwater overdraft, California signed the Sustainable Groundwater Management Act (SGMA) into law in



Results from a 2-year study suggest that applying winter runoff to Central Valley orchards in moderately drained to well-drained soils has minimal effects on yield, root production and light interception. *Photo*: David Doll.

2014. SGMA requires groundwater-dependent regions to combat the drop in groundwater levels by developing plans to balance pumping and recharge.

One promising approach in this effort is to transfer surplus surface water into groundwater aquifers during winter on agricultural lands (O'Geen et al. 2015). While this practice is relatively easy with annual crops that have a fallow period, this option has not been widely explored yet with perennial crops, in part due to concerns that prolonged soil saturation may damage crop root systems. A recent study on alfalfa in California demonstrated the feasibility of this approach in highly permeable soils (Dahlke et al. 2018). The large acreage of California's almond orchards and the available water distribution infrastructure used to support it could potentially facilitate groundwater reservoir recharging in these orchards during winter, but it is not known what potential effects flooding might have on the trees' aboveground growth and production. It is also not known what effect flooding might have on the trees' root systems. In particular, there may be concerns with exposing the perennial roots to potentially damaging low-oxygen conditions when orchards are kept saturated (Kozlowski 1997). Responses of roots to groundwater recharge are important because roots play a vital role in water and nutrient uptake (Osmont et al. 2007). They also function as anchors and storage organs, providing carbohydrates to restart aboveground development after the dormancy period ends (Tixier et al. 2019).

To evaluate the impact of winter flooding on almond root growth, canopy development, whole-plant water status and yield, we conducted field experiments in two commercial almond orchards in California's Central Valley, one with highly permeable soil and one with moderately permeable soil. Because our recharge treatments occurred during the dormant season, we hypothesized that almond trees would be able to tolerate saturated or nearly saturated soil conditions during this period without negative effects on root growth, water status or yield. In California, there are over 5 million acres of soils with Agricultural Groundwater Banking Index (SAGBI) ratings of excellent, good and moderately good (O'Geen et al. 2015). Most of these soils are on the east side of the Central Valley; the findings from this study will benefit those areas, with implications for the practice of groundwater recharge in dormant orchards.

Experimental sites and design

We conducted field experiments simultaneously in two almond orchards, one near Delhi, the other near Modesto (fig. 1), from December 2015 to October 2017. The orchard near Delhi (37 24'16 N, 120 47'20 W) was established in 2000 with alternating rows of Butte and Padre varieties on Nemaguard rootstock. Trees are spaced 18 feet apart with 22 feet between rows. The soil type at this site is Dune Sand with a SAGBI rating

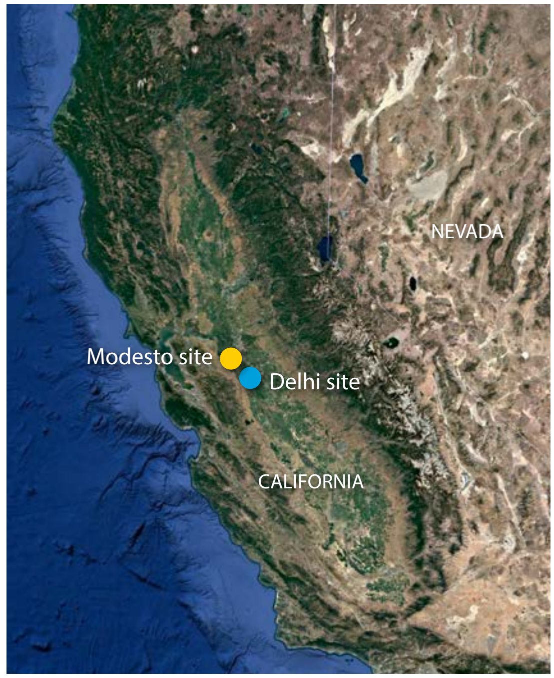


FIG. 1. Location of field sites in Delhi (Merced County) and Modesto (Stanislaus County), California. *Image:* Google Earth.

of "excellent". The second orchard, near Modesto in Stanislaus County (37 36'30 N, 121 04'20 W), was established in 1996 with alternating rows of 50% Nonpareil and 25% each of Monterey and Sonora varieties on Nemaguard rootstock. Trees are spaced 21 feet apart with 22 feet between rows. The soil in this orchard is classified as Dinuba Fine Sandy Loam, with a SAGBI rating of "moderately good" (O'Geen et al. 2015). Soil stratigraphy at each field site is illustrated in the online technical appendix. We obtained precipitation data from stations #71 Modesto and #206 Denair II of the California Irrigation Management and Information System (CIMIS; https://cimis.water.ca.gov).

Capturing stormwater runoff and potentially banking it in groundwater through winter irrigation in almond orchards might be a feasible method to reduce groundwater overdraft in California.

At each site, we applied recharge and control treatments to different sections of the same orchard block. At Modesto during the growing season the orchard is basin flood–irrigated approximately every 3 weeks using surface water provided by the local irrigation district. During January 2016 and January 2017, we applied 6 inches of water weekly (a total of 24 inches each month) to nine contiguous recharge treatment rows via flood irrigation, using city stormwater runoff captured by the Modesto Irrigation District and rerouted to irrigation canals. We measured root dynamics and stem water potential in five randomly selected trees from three center Nonpareil rows, and we measured yield and light interception for all Nonpareil trees in the treatment block.

At Delhi, we chose five rows, each with 32 trees (alternating Butte/Padre), for our experiment. During the active growing season, the grower irrigates these rows using micro-sprinkler irrigation. During our study, from December 2015 to mid-January 2016 and again during January 2017, we applied 8 inches of water to the first 10 trees in each row in three separate events (24 inches total per season) via flood irrigation with pumped up local groundwater. We used the last 12 trees in each row for control measurements. As in the Modesto orchard, we measured root dynamics, stem water potential, yield and light interception on five randomly selected trees; we selected trees for this purpose from the center row (Butte). Dates and amounts of groundwater recharge events in both sites are shown in table 1.

Measurements and data analyses

During our 2-year study, we measured soil water content for each treatment at each experimental site at 10-minute intervals at depths of 6 inches, 18 inches and 40 inches using GS3 soil-moisture sensors (Decagon Devices, Pullman, Wash.). We measured stem water potential (Ψ_{stem}) of bagged leaves in the active growing season and twigs in the dormant season bi-weekly. We measured root-growth dynamics from minirhizotron root images that we collected every 3 weeks using a portable CID root imager (CID Bio-Science, Camas, Wash.). (We installed clear root observation tubes to a 2-foot soil depth at an angle of 60° and inserted swimming pool noodles to prevent temperature gradients. We capped and covered the tubes with sand-filled bags to prevent them from flooding and/or floating away.) We hand-traced roots in the images using RootFly software (Clemson University), and from the tracings we calculated total lengths of new roots and of disappeared/dead roots through time. We measured canopy light interception (i.e., photosynthetically active radiation below the canopy) during the growing seasons in 2016 and 2017 using methods described in Zarate-Valdez et al. (2015). We measured yield at harvest in 2015 (pre-treatment) and again in 2016 and 2017. We used a t-test to determine whether there was a significant difference between the means of two treatment groups at a significance level of $P = 0.05$. More details on measurements and data calculation can be found in the technical appendix.

TABLE 1. Dates of groundwater recharge events and amount of applied water for each event during the winters of 2015–2016 and 2016–2017 at Delhi and Modesto

Season	Delhi		Modesto	
	Date	Irrigation amount	Date	Irrigation amount
		inches		*inches*
2015–2016	12/23/15	8	1/4/16	6
	12/29/15	8	1/11/16	6
	1/12/16	8	1/19/16	6
			1/25/16	6
2016–2017	1/13/17	8	1/9/17	6
	1/19/17	8	1/16/17	6
	1/26/17	8	1/23/17	6
			1/30/17	6

Soil water content in response to winter watering

We observed that the deep percolation rate of applied water in the sandy soil of Delhi was higher than the deep percolation rate in the sandy loam soil at Modesto (table 2). This suggests that soil permeability is one of the major factors determining the efficiency of groundwater recharge in winter. Natural precipitation during the second season of our study (October 2016 to April 2017) was significantly higher than it was during the first season at both Delhi (35% increase) and Modesto (26% increase) (table 2). This explains the greater deep percolation rate of applied water in both sites in 2017 compared to 2016 (6% and 15% increases at Delhi and Modesto, respectively).

Soil moisture sensors in Modesto showed that soil water content at this site depleted more quickly in deep soil (at 3.3-foot depth) than in shallow soil (at 0.5-foot

TABLE 2. Water inputs (precipitation and applied water for groundwater recharge) and estimated deep percolation and loss of applied water to soil storage from October to April of 2015–2016 and 2016–2017 at Delhi and Modesto

Site	Precipitation	Applied water	Total deep percolation	Deep percolation from rainfall	Deep percolation of applied water		Loss of applied water to soil storage	
	inches	*inches*	*inches*	*inches*	*inches*	*percentage*	*inches*	*percentage*
2016								
Delhi	12.94	26.15	29.09	4.79	24.31	93	1.84	7
Modesto	9.91	24.00	21.90	2.55	19.35	81	4.65	19
2017								
Delhi	17.44	25.80	33.03	7.43	25.60	99	0.20	1
Modesto	12.46	24.00	27.94	4.78	23.16	96	0.84	4

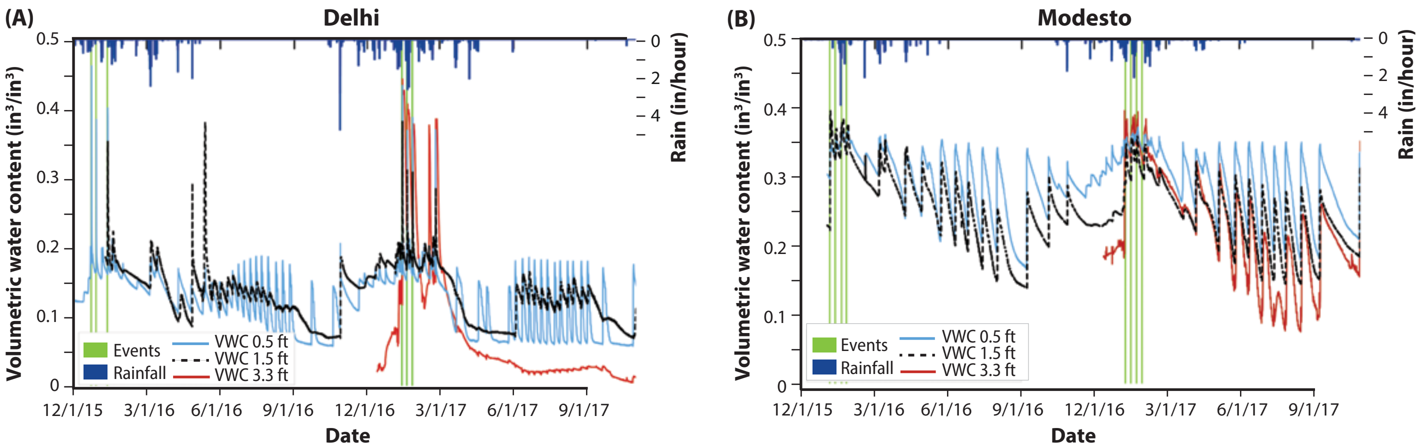


FIG. 2. Volumetric water content (VWC, in^3/in^3) for winter-watered almond orchards at (A) Delhi and (B) Modesto, measured at 0.5 ft (15 cm; blue solid lines), 1.5 ft (45 cm; black dashed lines) and 3.3 ft (100 cm; red solid lines). Blue bars represent the daily precipitation amount (inches/hour); green bars represent groundwater recharge events.

and 1.5-foot depths) at the beginning of 2017 (fig. 2), suggesting that, at Modesto, deeper layers have greater hydraulic conductivity (supplementary figs. 1 and 2 in the technical appendix).

Soil texture significantly influenced residence time of the water as well as deep percolation rates. Maximum soil water content at 1.5-foot depth after one recharge event was reached much more quickly in the sandy soil at Delhi (1 hour) than in the fine sandy loam at Modesto (more than 24 hours, fig. 3). Root-zone residence time (RZRT) of flood water, defined as the length of time it takes for soil water content to return to pre-flooding conditions after each event of groundwater recharge, was much longer at the Modesto site (6 inches of water applied per event, RZRT > 72 hours) than at the Delhi site (8 inches applied per event, RZRT < 24 hours).

Water status and root growth

We found no negative effects of groundwater recharge on tree water status. Ψ_{stem} during winter and in early spring was at or higher than the baseline for all trees at both field sites in both years (fig. 4). In both years the last winter groundwater recharge event took place in late January, and the introduced water stayed in the root zones no more than a week. At this time of year the trees have not leafed out yet and thus we would not expect any direct effects of water added on the physiology of the tree unless the tree was water stressed or the root system was negatively affected by saturated conditions in the root zone. We found no evidence of increased root death or decreased root production in the months immediately after the recharge events were applied in either year (table 3, January–March). However, in 2016 we found less negative in-season Ψ_{stem} for trees in plots where winter water for recharge was applied compared to the control (no extra water applied) at Delhi. This was likely due to other factors than the winter recharge treatment. At

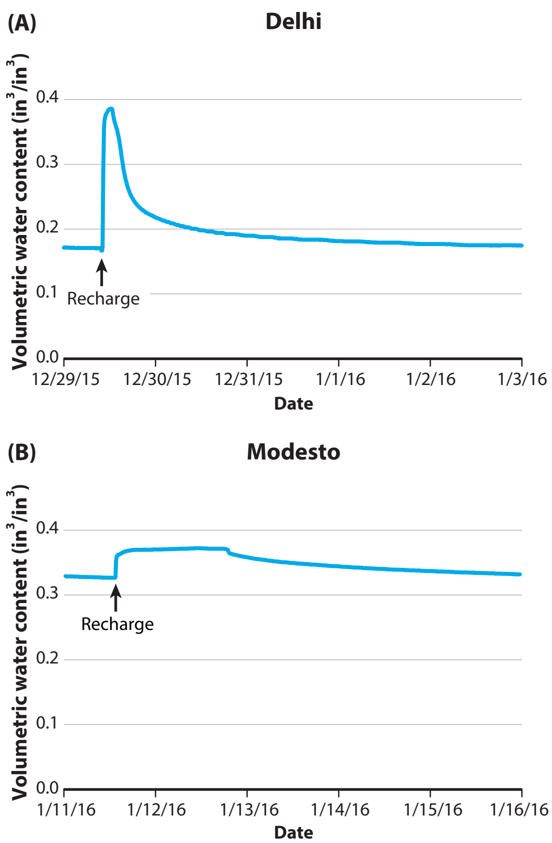


FIG. 3. Changes in volumetric water content (VWC, in^3/in^3) at 1.5 ft (45 cm) soil depth in response to a single flood event (black arrows) at (A) Delhi and (B) Modesto. During each groundwater recharge event, 8 inches of water were applied at Delhi, and 6 inches at Modesto.

Delhi the recharge plots had a deeper layer of sandy soil in the recharge plot, which may have allowed deeper root growth under the high frequency summer irrigation regimen typical of orchards located on sandy soils.

Adding winter water for groundwater recharge showed no adverse effects on new root production at either site (tables 3 and 4). Almond trees produce most new roots around the stage of nut development, from April to June (see example in fig. 5). At Delhi, we found no significant increase in total length of new roots in winter-watered trees in the first months after

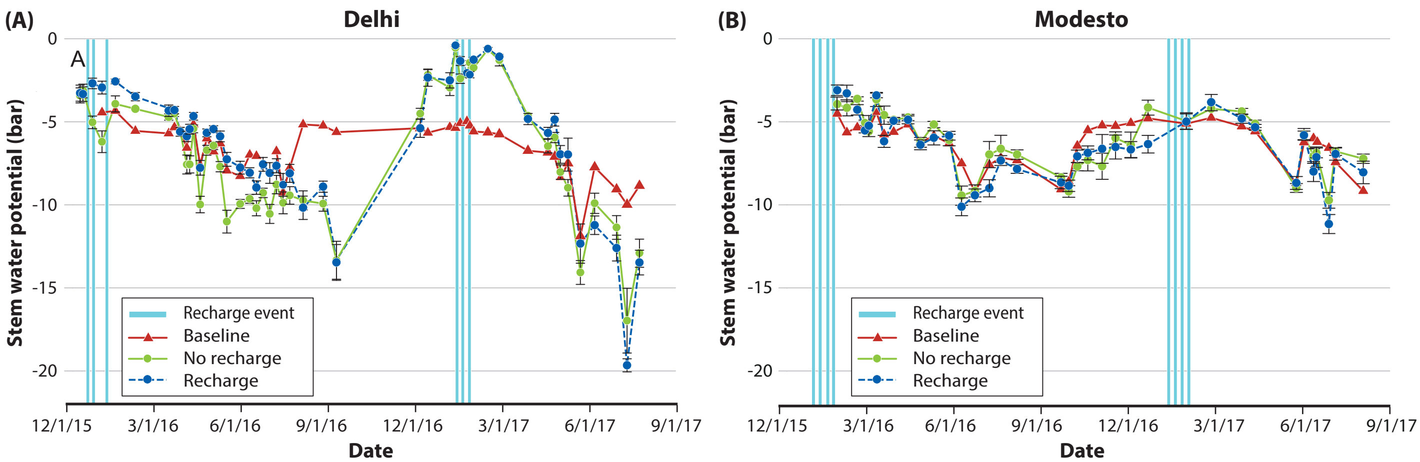


FIG. 4. Stem water potential (Ψ_{stem}, bar) of irrigated almond trees (blue circles) and nonirrigated trees (green circles) for winter groundwater recharge events in (A) Delhi and (B) Modesto in 2016–2017. Baseline (red triangle) was the expected water potential for well-watered trees based on weather conditions during the measurement period. Blue bars represent the events of groundwater recharge. Error bars represent standard error (n = 5).

TABLE 3. Seasonal changes in total lengths of new and dead roots at Delhi with and without winter groundwater recharge treatment

Year	Time period	Total length of new roots (in/ft$^2_{tube\ surface}$)		Total length of dead roots (in/ft$^2_{tube\ surface}$)	
		No recharge	Recharge	No recharge	Recharge
2016	January–March	6.99 ± 2.56	7.52 ± 3.81	0.56 ± 0.20	2.49 ± 2.13
	April–June	20.00 ± 11.57	13.08 ± 3.15	**7.52 ± 2.14**	**1.59 ± 0.69**
	July–September	8.08 ± 2.93	8.07 ± 2.87	1.95 ± 1.20	4.22 ± 1.62
	October–December	0.97 ± 0.40	4.70 ± 1.93	4.08 ± 2.98	2.31 ± 0.99
2017	January–March	2.10 ± 1.39	1.74 ± 1.02	4.98 ± 1.74	11.09 ± 4.43
	April–June	9.15 ± 4.49	3.97 ± 0.51	4.61 ± 1.59	8.17 ± 2.02
	July–September	5.63 ± 2.60	4.20 ± 2.01	8.89 ± 3.45	6.23 ± 1.26
	October	0.03 ± 0.03	0.60 ± 0.21	4.20 ± 1.22	3.72 ± 1.10

Numbers represent mean ± standard error; bold numbers indicate a statistically significant difference (P < 0.05) between treatments.

TABLE 4. Seasonal changes in total lengths of new and dead roots at Modesto with and without winter groundwater recharge treatment

Year	Time period	Total length of new roots (in/ft$^2_{tube\ surface}$)		Total length of dead roots (in/ft$^2_{tube\ surface}$)	
		No recharge	Recharge	No recharge	Recharge
2016	January–March	0.81 ± 0.51	4.84 ± 4.13	0.00 ± 0.00	0.00 ± 0.00
	April–June	12.90 ± 2.90	15.82 ± 5.16	3.29 ± 1.47	0.60 ± 0.30
	July–September	2.25 ± 0.50	3.21 ± 0.60	9.05 ± 2.63	3.86 ± 0.89
	October–December	0.93 ± 0.55	3.14 ± 0.83	1.87 ± 0.91	4.28 ± 1.22
2017	January–March	2.99 ± 0.84	3.86 ± 1.12	2.21 ± 0.67	4.96 ± 1.78
	April–June	4.47 ± 2.02	3.97 ± 0.35	3.06 ± 0.20	5.12 ± 0.79
	July–September	0.52 ± 0.26	0.90 ± 0.26	2.93 ± 1.45	5.29 ± 1.02
	October	0.19 ± 0.19	0.16 ± 0.05	0.82 ± 0.44	1.97 ± 0.65

Numbers represent mean ± standard error.

the recharge treatment was applied (January–March), yet there was a trend to lower April–June new root length production in the recharge treatment in both years (table 3). In the Modesto orchard, trees that received extra winter water showed a tendency to produce more new roots in the first quarter (January–March) of each treatment year (table 4), especially in 2016, which had low winter rainfall. These results indicate that winter irrigation does not have a statistically significant impact on root development in highly

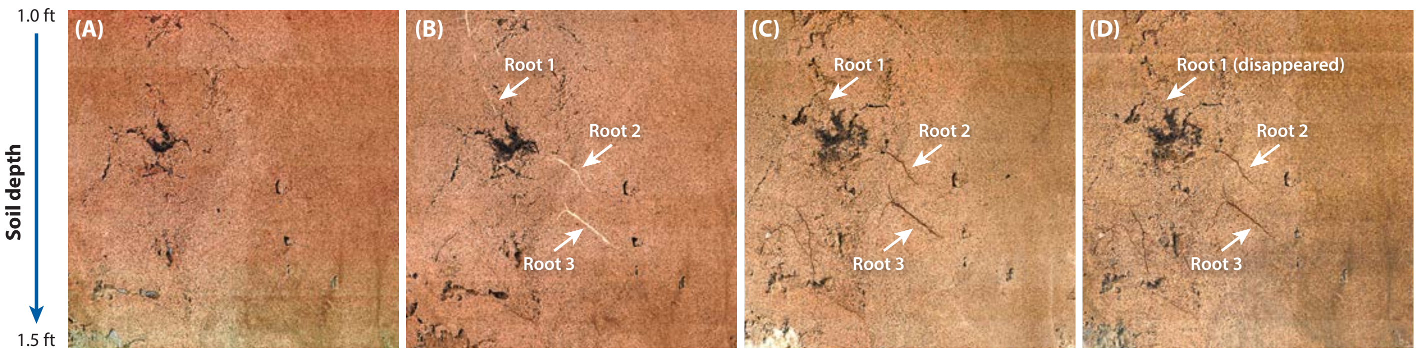


FIG. 5. An example of root growth dynamics at the Delhi site. Raw root images were taken at soil depths between 1 and 1.5 feet (30–45 cm) by using the CI-600 root imager (CID Bio-Science) in (A) February, (B) May, (C) September and (D) December of 2016. *Photos:* Paul Martinez.

permeable sandy soils or moderately permeable soils (e.g., sandy loam).

Standing root length is the net result of both new root length produced and root length that has died. When studying the impact of a treatment on root death, it is important to keep in mind that roots first need to be produced before they can die. Thus, a high root length that died can be either the result of high production in a previous month or the result of accelerated root death (reduced lifespan of produced roots). At Delhi, reduced length of dead roots in the recharge treatment in April–June 2016 reflects the lower new root production in that same period (keeping standing root length the same). In 2016 at Modesto, however, we had considerably higher new root length production through June in the recharge treatment but this was matched with much reduced dead root length production, thus suggesting that the lifespan of the roots was longer in the winter recharge plots. We did not find this in 2017. An extended lifespan reduces the ability of roots to take up water and soil nutrients (Volder et al. 2004). This pattern was not repeated in 2017, suggesting that variations in climate or soil conditions between the plots and years, not recharge treatments, could explain the results. Significantly higher precipitation in 2017 (table 2) increased soil water availability for root growth both in the control and in the treatment plots, thus minimizing any potential positive effects of winter irrigation.

Canopy light interception and yield

Groundwater recharge in winter showed minimal effects on canopy development and nut production; canopy light interception and yield were similar between treatments during each year at both field sites (table 5). Both sites had slight decreases in percentage of canopy light interception, indicating a reduced canopy size across treatments (with and without groundwater recharge) in the wet year of 2017 compared to the dry year of 2016. This is to be expected based on patterns of spur dynamics; more spurs die in a dry year, thus leading to reduced canopy size in the following year (Lampinen et al. 2011).

While annual yield at Modesto was fairly consistent over the two years of our study, we observed substantial annual variation in yield at Delhi. The year 2016 was a low-producing year in both winter-watered and control treatment blocks at Delhi, while 2017 was a higher-producing year, especially in the recharge treatment block (table 5). However, there was also greater yield in this same block in 2015, the year prior to the application of winter recharge (+46% and +41% greater production in the recharge block in 2015 and 2017, respectively). The higher yields in 2015 (pre-treatment) and 2017 in the recharge block at Delhi support the idea that trees there may have deeper root systems, which help maintain high nut production in the years following a dry year by enabling greater spur survival (Lampinen et al. 2011). At Delhi, the soil profiles between the recharge treatment and the control block were sufficiently

TABLE 5. Canopy light interception (%) and almond yield (lb/acre) for blocks grown with and without winter groundwater recharge at Delhi and Modesto in 2016 and 2017

	Canopy light interception (%)		Yield (lb/acre)					
	2016	2017	2015*	Percentage	2016	Percentage	2017	Percentage
Delhi								
No recharge	72.0	65.3	2,415	100.0	1,575	100.0	2,202	100.0
Recharge	75.8	65.4	3,535	146.0	1,393	88.0	3,108	141.0
Modesto								
No recharge	88.8	75.1	3,360	100.0	3,291	100.0	2,982	100.0
Recharge	85.2	77.2	3,425	102.0	3,129	95.0	2,985	100.0

* Results for 2015 reflect pre-experiment conditions.

different (see technical appendix) that this is a more likely explanation than the recharge treatment *per se*.

Thus, we found no positive or negative effect of adding water for winter recharge on yield at either Delhi or Modesto. It is possible that younger almond orchards (i.e., those less than 15 years old) might have different responses to winter recharge treatment, which is a possibility that needs to be investigated in future studies.

Minimal negative effects, potential benefits of winter watering

Capturing stormwater runoff and potentially banking it in groundwater through winter irrigation in almond orchards might be a feasible method to reduce groundwater overdraft in California. In our study, over 90% of the winter-applied water percolated past the root zone (2-foot depth) in the sandy soil at Delhi and 80% percolated past the root zone in the fine sandy loam at Modesto (table 5). Our data show that this watering had minimal effects on yield, root production and light interception in both almond orchards. However, as we added extra water for recharge purposes to only one block per treatment at each site, we cannot separate the effects of differences across blocks from the effects of the recharge treatments, and thus we cannot firmly conclude that winter watering has no negative impacts on almond orchards. More rigorous and longer-term studies are necessary to confirm this low risk and perhaps explore potential horticultural advantages of winter irrigation in *Prunus* spp. orchards at different ages.

The opportunity to flood almond orchards during the dormant season may only be feasible during years when winter rains are above normal. More studies are needed to evaluate the impact of applying water for recharge purposes later, in the spring, when more surface water becomes available in most parts of the Central Valley. This is when the roots and shoots are actively growing (after blooming or during the fruit development stage, April–May), and trees that are actively growing are much more susceptible to the negative effects of low oxygen conditions in the soil (Kreuzwieser and Rennenberg 2014). In addition, due to orchard growing practices and fertilizer applications, this period is likely much less suitable for groundwater recharge (Duncan et al. 2019), as it carries an additional risk of leaching nitrates and other pollutants into the groundwater and there is a need to regularly move heavy equipment through the orchard.

Lastly, efficiency of groundwater recharge and its effects on the growth of almond trees are influenced by rootstock, soil type and other factors that affect water percolation. In order to prevent unintended tree loss, growers need to carefully consider these factors when adopting the strategy of groundwater recharge in almond orchards. CA

X. Ma was Postdoctoral Researcher, Department of Plant Sciences, UC Davis, and is now Assistant Professor, Institute of Agricultural Resources and Environment, Jiangsu Academy of Agricultural Sciences, China; H. Dahlke is Professor, Department of Land, Air and Water Resources, UC Davis; R. Duncan is Farm Advisor, UC Cooperative Extension, Stanislaus County; D. Doll is Orchard Operations Manager, Rota Unica, Evora, Portugal; P. Martinez is Graduate Student, B. Lampinen is UC Cooperative Extension Specialist, and A. Volder is Professor, Department of Plant Sciences, UC Davis.

We wish to thank the Almond Board of California for funding this project through grants 15-17 PREC9 and 20 PREC10. We also thank the Modesto Irrigation District, cooperating growers Nick Blom (Modesto) and Glen and Craig Arnold (Delhi), as well as help and advice from Dan Rivers, Sam Metcalf, Ken Shackel, Loreto Contador, Colin Fagan, Rebecca Scott, Elexia Koch, Seanna McLaughlin and Cristina Prieto Garcia. This research was partially supported by the California Agricultural Experiment Station with funding from the Hatch Act through the USDA National Institute of Food and Agriculture.

References

[CDFA] California Department of Food and Agriculture. 2020. California Agriculture Statistics Review 2019-2020. www.cdfa.ca.gov/Statistics/PDFs/2020_Ag_Stats_Review.pdf (accessed Sep. 6, 2021).

[CADWR] California Department of Water Resources. 2019. Data and Tools. https://water.ca.gov/Programs/Groundwater-Management/Data-and-Tools (accessed Jan. 28, 2020).

Dahlke H, Brown A, Orloff S, Putnam D. 2018. Managed winter flooding of alfalfa recharges groundwater with minimal crop damage. Calif Agr 72(1):1–11. https://doi.org/10.3733/ca.2018a0001

Duncan R, Gordon P, Holtz B, et al. 2019. Sample costs to establish an orchard and produce almonds. UC Agricultural Issues Center. UC Davis. https://coststudyfiles.ucdavis.edu/uploads/cs_public/79/86/79863d8a-8f8b-4379-91c9-0335e20a2dd2/2019almondssjvnorth.pdf (accessed Aug. 5, 2020).

Kozlowski T. 1997. Responses of woody plants to flooding and salinity. Tree Physiol 17(7):490. https://doi.org/10.1093/treephys/17.7.490

Kreuzwieser J, Rennenberg H. 2014. Molecular and physiological responses of trees to waterlogging stress. Plant Cell Environ 37:2245–59. https://doi.org/10.1111/pce.12310

Lampinen B, Tombesi S, Metcalf S, DeJong T. 2011. Spur behavior in almond trees: Relationships between previous year spur leaf area, fruit bearing and mortality. Tree Physiol 31:700–6. https://doi.org/10.1093/treephys/tpr069

O'Geen A, Saal M, Dahlke H, et al. 2015. Soil suitability index identifies potential areas for groundwater banking on agricultural lands. Calif Agr 69(2):75–84. https://doi.org/10.3733/ca.v069n02p75

Osmont K, Sibout R, Hardtke C. 2007. Hidden branches: Developments in root system architecture. Annu Rev Plant Biol 58(1):93–113. https://doi.org/10.1146/annurev.arplant.58.032806.104006

Tixier A, Gambetta G, Godfrey J, et al. 2019. Non-structural carbohydrates in dormant woody perennials; the tale of winter survival and spring arrival. Front For Glob Change 2:18. https://doi.org/10.3389/ffgc.2019.00018

Volder A, Smart D, Bloom A, et al. 2004. Rapid decline in nitrate uptake and respiration with age in fine lateral roots of grape: Implications for root efficiency and competitive effectiveness. New Phytol 165(2):493–502. https://doi.org/10.1111/j.1469-8137.2004.01222.x

Zarate-Valdez J, Metcalf S, Stewart W, et al. 2015. Estimating light interception in tree crops with digital images of canopy shadow. Precis Agric 16:425–40. https://doi.org/10.1007/s11119-015-9387-8

RESEARCH ARTICLE

Fine-tuning fertilizer applications in organic cool-season leafy green crops can increase soil quality and yields

Organic vegetable growers can use soil nitrate tests to better understand how much organic fertilizer to apply.

by Richard Smith, Michael Cahn, Tim Hartz, Daniel Geisseler and Patricia Love

Online: https://doi.org/10.3733/ca.2022a0010

Abstract

Organic cool-season vegetable growers on the Central Coast face challenges in applying nitrogen (N) to balance yields with new environmental regulations. It is hard to time fertilizer applications while calculating N mineralization of soil organic matter and organic fertilizers to plant-available N. Organic fertilizers with high phosphorus (P) to N ratios may elevate P levels and harm surface water quality. In this study, we evaluated (1) mineralization of soil organic matter and fertilizers, (2) effectiveness of residual soil nitrate-N tests and (3) long-term impacts of organic fertilizers on P levels and soil microbial activity. We found that mineralization of N from soil organic matter provided limited N to leafy green vegetables. Soil tests were more reliable in heavier than sandier soils. Application rates of 4-4-2 were calculated to meet N demands, resulting in an oversupply of P. However, only 9% to 17% of fertilizer P solubilized without elevating available soil P levels. While it's difficult for organic vegetable growers to use cover crops, organic fertilizers increased carbon levels, resulting in higher levels of soil microbial activity.

Central Coast California growers produced $766 million of cool-season organic produce in 2019. These crops included lettuces, cole crops, spinach, strawberries and spring mixes (baby lettuces and kale, mizuna and arugula). Large-scale organic vegetable production on the Central Coast has been steadily increasing over the past 20 years due to rising consumer demand and optimal growing conditions. While there is basic information on organic soil fertility for vegetables (Gaskell et al. 2000; Gaskell and Smith 2007), specific soil fertility information for cool season vegetables is scarce. Organic growers have the added challenge of complying with recent nitrogen (N) use limits put out by the Central Coast Regional Quality Control Board.

A key practice used by conventional growers to improve the uptake efficiency of applied N fertilizer is measuring levels of residual soil nitrate-N and adjusting fertilizer application rates accordingly (Hartz et al. 2000). In organic vegetable production, however, soil nitrate tests are rarely used to guide fertilizer applications due to the uncertainty of how much (and when)

Results from a UC Cooperative Extension study suggest that large-scale organic vegetable operations on the Central Coast that seldom use cover crops or compost can benefit from the extensive use of organic fertilizers to stimulate soil microbial activity.

N is mineralized from soil organic matter and organic fertilizers. Soil tests for residual soil nitrate-N are most accurate when done immediately prior to a fertilizer application. However, the time lag in the release of nitrate-N from organic fertilizer adds a level of uncertainty to the use of nitrate-N tests, especially if residual soil N is lost to leaching before the crop can utilize it. It is critical to understand the release characteristics of organic fertilizers to determine how organic N fertilizer applications can be adjusted and made more efficient. It is also important to understand background levels of mineralization of N from soil organic matter, as well as the quantity of nitrate-N that is supplied by irrigation water, both of which can provide significant amounts of N for crop growth.

This project was conducted over 4 years with large-scale organic cool-season vegetable operations in Monterey County, which typically farm 300 to 1,000 acres and supply nationwide markets. Nitrogen mineralization from soil organic matter and organic N fertilizers were measured, and the utility of nitrate-N testing for residual soil N to adjust fertilizer applications was evaluated. The role of organic phosphorus (P) fertilizers in elevating soil P levels was examined and compared with neighboring conventional vegetable fields. In addition, the impact of the long-term use of organic fertilizers on soil microbial activity and organic matter was evaluated.

Conventional and organic fields

Trials were conducted with cooperating growers in conventional and organic romaine lettuce, spinach, spring mix and broccoli fields. Twenty experiments were conducted in 2016 and 2017 in Monterey County with the goal of evaluating N mineralization from soil organic matter in laboratory and in-field evaluations (table 1). Four composite preplant soil samples up to 12 inches deep were collected at each organic site and at a neighboring conventional farm with the same soil

TABLE 1. Twenty soil mineralization sites: details on crops, soil, fertilizer and yield

Site	Years certified organic	Prior crop*	Crop*	Days to harvest	Soil texture†	Initial soil mineral N	Soil N mineralization‡	Mean crop N uptake§	Total crop N uptake¶	Fertilizer N applied	Fertilizer N available (estimated)#	% yield increase with fertilization**
						ppm	*lb/ac/day*	*lb/ac/day*	*lb/ac*	*lb/ac*	*lb/ac*	
1	9	Spinach	Spinach	28	loam	14.3	—	1.62	45	210	126	17
2	14	Spinach	Spinach	22	s loam	27.0	1.19	3.22	71	120	36	16
3	5	Mizuna	B. lettuce	28	loamy s	21.1	0.31	2.45	69	90	24	37
4	20	W. fallow	B. lettuce	31	c loam	18.8	0.50	1.25	39	120	36	31
5	14	B. lettuce	Broccoli	64	s loam	14.0	0.83	5.88	376	437	219	62
6	2	B. lettuce	B. lettuce	31	c loam	14.2	0.32	1.47	46	160	72	21
7	2	B. lettuce	B. chard	25	si loam	24.1	0.10	4.29	107	160	72	17
8	5	B. lettuce	Romaine	39	loamy s	6.9	0.36	1.77	69	160	54	93
9	14	Romaine	Romaine	67	s loam	22.2	—	1.12	75	360	204	0
10	2	B. lettuce	Romaine	41	c loam	28.7	0.40	2.27	93	160	72	18
11	10	W. fallow	B. kale	31	loam	17.6	0.44	1.56	48	120	62	85
12	10	W. fallow	Romaine	51	loam	15.6	0.48	2.29	117	160	51	10
13	15	W. fallow	Spinach	29	c loam	26.3	0.25	0.96	28	160	53	210
14	15	Romaine	Broccoli	69	s loam	49.5	1.40	4.73	326	451	163	21
15	21	W. fallow	B. lettuce	35	c loam	34.0	0.30	1.39	49	150	53	20
16	15	Cauliflower	Romaine	56	s loam	47.1	0.59	2.04	114	440	159	18
17	6	Romaine	B. lettuce	27	s loam	12.4	1.10	2.44	66	148	49	44
18	6	Romaine	B. lettuce	27	loamy s	8.0	0.61	2.12	57	148	49	37
19	3	Romaine	B. kale	22	si loam	30.7	1.90	3.85	85	160	60	0
20	3	Spinach	B. kale	25	c loam	28.9	0.30	2.26	57	80	41	43
Mean						**23.1**	**0.63**	**2.45**	**97**	**199**	**83**	**40**

* B. lettuce = baby lettuce; B. chard = baby Swiss chard; B. kale = baby kale; W. fallow = winter fallow.

† s loam = sandy loam; loamy s = loamy sand; c loam = clay loam; si loam = silt loam.

‡ Laboratory estimates of net N mineralization from soil organic matter by intact core method (10 weeks at 59°F and optimal soil moisture content; Miller et al. 2018).

§ Total crop uptake from fertilized plots divided by crop cycle days.

¶ Biomass N in fertilized plots.

Estimates of net N mineralization from applied fertilizer based on laboratory mineralization of the fertilizer type used at the site.

** Percent increase in yield of grower standard fertilizer treatment over untreated control.

type and similar crop mix. The soils were analyzed for organic matter (loss on ignition), total N and C (combustion), bicarbonate extractable P (Olsen P), total P (acid digestion and ICP-AES analysis), permanganate extractable carbon (POXC) (Culman et al. 2012), water extractable N (WEON) and C (WEOC) (Haney et al. 2012), and fluorescein diacetate (FDA) hydrolysis (Green et al. 2006).

In each field, three replicates of unfertilized plots versus plots fertilized with the grower standard N application were established to measure the level of response to N fertilization. Plots were 6.7 feet wide by 10 feet long. In sprinkler-irrigated fields, three 1-liter water containers with 7-inch diameter funnels were installed; in drip-irrigated fields, in-line flow meters were installed in each plot to measure total applied water. Nitrate-N content of the water was measured, and total N applied in the irrigation water was calculated. Soil samples to 12 inches deep were collected weekly from each plot and were extracted with 2N KCl, to measure mineral N. Crop biomass and biomass N were measured at crop maturity.

In-field mineralization measurements were made on the unfertilized plots. Crop plants were removed, and the soil was covered with plastic mulch, in order to measure N mineralization from soil organic matter without the confounding factors of crop N removal and nitrate-N leaching by irrigation water. The plastic mulch was applied after the germination water wetted the soil. The soil remained at or near field capacity over the crop cycle — in other words, in a moist (but not saturated) condition, able to mineralize nitrogen with no leaching. White on black plastic mulch was used, with the white side facing up to reflect light and minimize soil heating, and the black side facing down to inhibit weed growth. Soil nitrate-N was measured each week during the cropping cycle by collecting six soil cores, 12 inches deep from each plot. Soil temperatures were measured using soil moisture and temperature probes (Decagon 5TM) and a recorder (Decagon Em50). In-field estimates of soil mineralization were made from the mulched plots by subtracting nitrate-N levels at the beginning of the crop cycle from nitrate-N at the end.

Nitrogen mineralization in soil from each site was also determined in laboratory assays by incubating undisturbed soil cores collected at the beginning of the crop cycle (Miller et al. 2018). A 6-inch-long by 2-inch-diameter undisturbed core was taken in a plastic sleeve from each replicated plot. At the same time, several samples were taken with a soil probe to the same depth from the area surrounding the core sampling location. These samples were then homogenized and stored in a cooler on ice with the undisturbed cores until returned to the laboratory. The cores were incubated at 59°F for 10 weeks. During the incubation, the soil moisture content was maintained at 60% water holding capacity. Net N mineralization was calculated as the difference between the initial mineral N measured in the samples from the surrounding soil and the final mineral N in the cores.

Each of the 20 organic field sites was paired with a neighboring conventional field of the same soil type and growing a similar mix of crops. Composite soil samples were collected from four locations in each field, and each replicate was analyzed for factors shown in table 2. The mean values for each soil parameter in the organic and conventional fields were statistically compared using a pairwise Student's t-test.

Evaluation of N and P release

Replicated in-field evaluations of N and P release from organic dry fertilizers were conducted in romaine lettuce fields in 2016 and 2017. Two commonly used organic fertilizers were evaluated: 4-4-2 (a mixture of poultry manure and feather meal) and 12-0-0 (feather meal). Twenty grams of oven-dried organic fertilizer, in a 0.5-inch-thick layer, were placed in 4-inch-by-5-inch polypropylene mesh bags, which were placed on the soil surface (to simulate a topdressed application) or buried three inches deep (to simulate a soil-incorporated application). The bags were subject to wetting and drying cycles from the sprinkler irrigation used in the field. To determine N and P release from the fertilizer, four bags were removed each week, and their contents were oven-dried, weighed and analyzed for total N and P. The quantity of N and P released from the fertilizer was calculated by the difference in the weight loss during the field incubation.

Aerobic laboratory incubations of organic fertilizers commonly used by organic growers were conducted at UC Davis. The incubations simulated soil

Large-scale organic vegetable operations that seldom use cover crops or compost can benefit from the extensive use of organic fertilizers, which stimulate soil microbial activity.

TABLE 2. Comparisons of soil parameters of 20 paired organic and conventionally managed vegetable production fields

If	Organic matter	C total	N total	Mineralization from OM	P total	P Olsen	POXC	FDA	WEOC	WEON
	%	*%*	*%*	*lb N/ac/day*	*%*	*ppm*	*ppm*	*mg/kg/hr*	*ppm*	*ppm*
Conventional	2.010	0.990	0.116	0.4000	0.104	36.600	269.400	11.500	92.200	13.500
Organic	2.070	1.030	0.117	0.6000	0.091	38.600	307.500	18.600	105.500	17.000
P-value	0.739	0.710	0.919	0.1265	0.189	0.710	0.202	0.007*	0.098	0.003*

* Indicates a highly significant difference between the means.

incorporation of the fertilizers. The soil was collected from two organically managed fields and was air-dried and thoroughly mixed. Organic fertilizers were blended with these soils at approximately 100 mg N kg^{-1} soil, brought to field capacity water content, and incubated at 68°F. At 2, 4 and 8 weeks of incubation, subsamples were extracted with 2N KCl, and mineral N was determined. The rate of net N mineralization from the organic fertilizers was determined as the increase in mineral N over time minus that measured in the control (unfertilized) soil.

To evaluate the use of soil nitrate tests in guiding fertilizer N applications, four replicated on-farm fertilizer trials were conducted in 2018 and 2019 in conventional and organic spinach fields. Soil textures at the sites were clay loams, silt loams and loamy sands (table 3). Soil organic matter ranged from 1.18% to 3.47%. Each plot was one 80-inch bed wide by 30 feet long and was replicated four times in a randomized complete block design. The following treatments were included in each trial: untreated control, 50% of grower standard, and grower standard fertilizer application. At each site, fertilizer either was applied at listing or top-dressed at planting. Soil samples were collected from the top 12 inches of soil at planting and each week until harvest, and were analyzed for mineral N. Yield and crop N uptake were measured at crop maturity.

N available for crop growth

During the crop cycle, a portion of the N in soil organic matter mineralizes to plant-available forms. This study measured mineralization of nitrate-N from the soils at the 20 study sites. Laboratory incubations averaged 0.6 pounds (lb) N/acre (ac)/day (range 0.3 to 1.9), and in-field evaluations averaged 1.6 lb N/ac/day (range 0.3 to 3.3). In spite of efforts to prevent soil heating under the plastic mulch, soil temperatures were higher in the mulched plots (data not shown), which may have given higher estimates of mineralization than have been reported in the literature for similar soil types (Miller et al. 2018). The correlation between the in-field and laboratory incubations was low ($R^2 = 0.08$). As a result, we had greater confidence in the laboratory estimates of N mineralization (table 1) and used that data to estimate N mineralization in this study. Fast-growing leafy vegetables such as lettuce and spinach take up 3 to 4 and 5 to 6 lb N/ac/day, respectively, during the last half of the growth cycle (Bottoms et al. 2012; Heinrich et al. 2013). Average daily soil mineralization over all sites was 0.6 lb N/ac/day and average crop uptake was 2.5 lb N/ac/day (table 1), indicating that daily crop N demand was higher for these crops than the amount mineralized from soil organic matter.

Nitrate-N in irrigation water can be an important source of N for crop growth (Cahn et al. 2017). However, the average levels of nitrate-N in the irrigation water at these study sites was 11.1 parts per million (ppm), with only two sites > 48.0 ppm and the rest below 16.0 ppm. On average, the irrigation water supplied 15.6 lb N/ac over the crop cycle (data not shown), an important but small portion of the N needs of the crops in this study.

Residual soil nitrate-N left over from prior rotations also can be a significant source of N for crop growth. This is particularly true for the second or third crop of the season, where significant mineral N remains from prior crop residues and unused fertilizer N. A soil nitrate-N concentration of 20 ppm in the top foot of soil is equivalent to 70 to 75 lb of N/ac (at soil bulk densities of 1.3 to 1.4 grams/cm^3) and provides sufficient N to supply crop needs for 1 to 2 weeks (Breschini and Hartz 2002). The mean concentration of residual soil nitrate-N at all sites at the beginning of the crop cycle was 23.1 ppm (83 lb N/ac, table 1). However, nitrate-N is susceptible to losses due to excess irrigation, particularly during crop stand establishment. An average of 3.0 inches/ac of water was applied to establish the crops at the study sites, while crop water demand during this time was 1.9 inches/ac (based on estimates of evapotranspiration in Johnson et al. 2016). As a result, an average of 1.1 inches of water leached (ranged from 0.1 to 2.9 inches) during the first 7 to 10 days of the crop cycle. This leaching loss may explain why residual soil nitrate at the beginning of the crop cycle did not play a bigger role in providing crop N needs (fig. 1).

Nitrogen uptake by the crop at harvest ranges from 120 to 160 lb/ac for lettuce, 90 to 130 lb/ac for spinach, 60 to 70 lb/ac for baby lettuce, and 250 to 350 lb/ac for broccoli (Hartz 2020). Fertilizer N

TABLE 3. Fertilizer trial site, crop background and yield response to N applied at 4-4-2

Site no.	Years organic	First water	Days to harvest	Soil texture*	Soil organic matter	N applied at listing	N applied at planting	Total N	Soil nitrate-N at planting	Yield response to fertilizer†
					%	*lb/ac*	*lb/ac*	*lb/ac*	*ppm*	
1	4	June 6	30	c loam	3.47	80	80	160	33.0	No
2	4	Aug 24	27	loamy s	1.18	160	—	160	28.3	Yes
3	4	Sept 20	26	c loam	3.18	40	80	120	18.3	No
4	5	July 15	24	si loam	2.25	80	80	160	21.6	No

* c loam = clay loam; loamy s = loamy sand; si loam = silt loam.

† Significant difference between the grower standard fertilizer application and the 0% and 50% fertilizer treatments, $P < 0.05$.

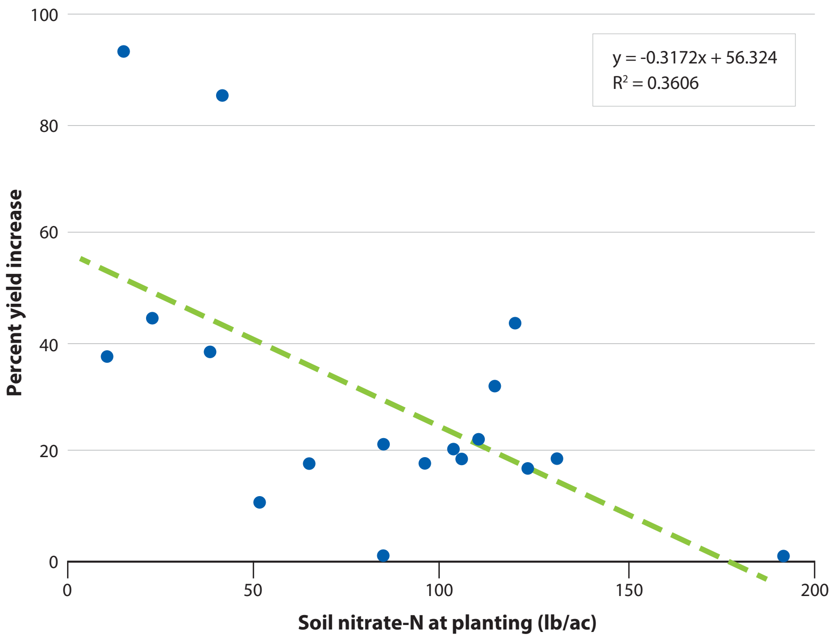


FIG. 1. Correlation (or relationship) between increase in crop yield and residual soil nitrate-N at planting. Increase in the yield of vegetables with fertilization in comparison to residual soil nitrate-N in the soil at the beginning of the crop cycle.

Polypropylene mesh bags containing organic fertilizer were placed on the soil surface to simulate a topdressed application. *Photo:* Richard Smith.

Bulk delivery of organic fertilizers. *Photo:* Richard Smith.

application rates are often at or above these N uptake values. However, as will be discussed below, the net amount of N released from organic fertilizers during the crop cycle is less than the total N content of the crop. Over all sites, the amount of N released from applied fertilizer was less than crop uptake: average crop uptake was 97 lb N/ac and the average net release of N from applied fertilizer was 83 lb N/ac (table 1). It appears that the other sources of N, such as residual mineral N, mineralization of N from soil organic matter, and nitrate-N in irrigation water, while modest, were sufficient to supplement fertilizer N to achieve economically viable yields.

A guide to fertilization

In conventional production systems, supplemental fertilizer applications can be made more precise by measuring residual soil nitrate-N levels immediately before fertilization and reducing fertilizer applications if there is adequate residual nitrate-N. If levels are ≥ 20 ppm nitrate-N, the need for fertilizer N can be reduced or eliminated (Breschini and Hartz 2002). Nitrogen in conventional fertilizers is soluble and immediately available to the crop. However, due to the time lag in the release of mineral N from organic fertilizers, soil nitrate-N tests in organic production cannot effectively be made immediately prior to fertilization. Rather, they need to be conducted early in the production cycle to make sure that the release of the mineral N from the fertilizer effectively meshes with crop N demand.

To better understand the utility of measuring soil nitrate-N to guide fertilizer applications in organic vegetable production, four fertilizer trials were conducted on spinach, a fast-growing and shallow-rooted crop. Given the short life cycle of the crop and the lag time in the release of mineral N from organic fertilizers, the only opportunity to test residual nitrate-N is at or before planting. All trial sites were the second or third crops of the season and had levels at or above 20 ppm nitrate-N at the beginning of the crop cycle (table 3). Treatments included 0%, 50% and 100% of the grower standard N application rate. There was no yield response to applied fertilizer on sites with heavy soils (sites 1, 3 and 4). Yields equivalent to the 100% grower standard were obtained with 0% or 50% of the standard fertilizer application rate. These results indicate that the quantity of residual N was adequate to supply crop needs. Site 2 had a loamy sand soil; even though residual soil nitrate-N levels were 28.1 ppm, fertilizer application rates of less than 100% of the grower standard resulted in lower yields. Leaching of the residual nitrate-N during the application of germination water was probably responsible for the lower yield in the reduced fertilizer plots at this site. The active roots of spinach are in the top 4 to 12 inches of soil (Heinrich et al. 2013), and maintaining nitrate in this narrow band of soil with sprinkler irrigation is challenging, especially on light soils. The results from these trials suggest that testing soil for residual soil nitrate-N can be useful in organic vegetable production. However, to estimate future availability of residual soil nitrate-N in soil, nitrate tests appear to be most useful for heavier soils. The tests are less useful for lighter soils, where nitrate-N can be leached by irrigation water before it is utilized by the crop. More studies are needed to better understand how to use nitrate tests to fine-tune fertilizer N applications in cool-season vegetables in organic systems.

Dry and liquid fertilizers

Laboratory incubations were conducted with a variety of dry and liquid organic fertilizers to understand how much and when N is mineralized. The fertilizers were mixed with moist soil and incubated at 68°F for 56 days. This technique simulates mineralization of N from fertilizer that is incorporated into moist soil. The quantity of N that mineralized from dry fertilizers ranged from 14.9% to 60.7%, and liquid fertilizers ranged from 51.8% to 68.7% of the initial N content of the fertilizer (table 4). Fertilizers with greater N concentrations released a greater proportion of N than fertilizers with lower N concentration. For example, 4-4-2 and 12-0-0 mineralized 41.4% and 60.7%, respectively, of their initial total N content over the 8-week incubation. Liquid fertilizers with N concentration equivalent to dry materials mineralized a greater proportion of their N and did so more quickly. Liquid fertilizers are typically made from high-N-content material, which is diluted with water during application.

In-field N release studies of dry organic fertilizers compared surface-applied and soil-incorporated applications of 4-4-2 and 12-0-0 over 55 days in a sprinker-irrigated lettuce field. Both fertilizers released a high proportion of their mineral N during the first 2 to 3

TABLE 4. 2017 Laboratory incubations: Percent of total nitrogen mineralized

Fertilizer type	Fertilizer analysis (N-P-K)	Days incubated at 68°F		
		14 days	28 days	56 days
Dry				
Poultry manure	2.5-2.0-2.5	4.4	7.7	14.9
Poultry manure and feather meal	4-4-2	29.7	31.7	41.4
Meat & bone meals	8-5-1	44.1	45.9	57.9
Feather meal, meat & bone meals, and potassium sulfate	10-5-2	44.7	51.5	57.2
Feather meal	12-0-0	50.5	57.7	60.7
Liquid				
Grain fermentation	2.5-2.0-1.0	30.6	31.6	51.8
Fish solubles, sugar molasses, beet extract	4-1-3	52.5	56.7	68.7

Lettuce mid growth cycle. *Photo:* Richard Smith.

weeks. Over 55 days, soil-incorporated applications of 12-0-0 released more N than 4-4-2, 86.0% versus 54.0% (fig. 2). Less N was released from surface applications of both materials than from soil-incorporated applications. This may be due to drying cycles between rounds of irrigation, which slow the mineralization process and reduce N release. Soil-incorporated fertilizer stays more consistently wet, allowing mineralization to continue uninterrupted.

Organic fertilizers have two distinct phases of release of mineral N: an initial rapid phase that occurs in the first 2 to 4 weeks after wetting of the material, during which time the labile forms of N mineralize, and a prolonged phase in which the recalcitrant components in the fertilizer slowly mineralize (Hartz and Johnstone 2006). In these trials, organic fertilizers did not mineralize all of the N that they contained. The unmineralized portion of organic fertilizers is recalcitrant and becomes part of the soil organic N pool, which slowly becomes available over subsequent years. As a result, the leaching potential of recalcitrant N in organic fertilizer is low. Ag Order 4.0, developed by the Central Coast Regional Water Quality Control Board, uses the quantity of applied N ("A" value) and the quantity of N removed by the crop ("R" value) as the basis for N fertilizer use regulations. The results from this study and Lazicki et al. (2020) underscore the need to consider mineralized net N rather than the total amount applied from organic fertilizers when calculating the "A" value for organic fertilizer.

Organic fertilizers release P

The risk of elevating P levels is a common concern in organic agriculture due to the use of P-rich composts and fertilizers (Maltais-Landry et al. 2016). Excess soil P can be transported in runoff from storms or irrigation to surface waters and can cause water quality impairment by eutrophication. The high P:N ratio of organic fertilizers such as 4-4-2 result in high application rates of P, because the material is applied at rates that satisfy the N needs of the crop, but are excessive for the crop's P needs.

This study provided preliminary evaluations of the impact of the use of 4-4-2 on P buildup in Salinas Valley soils. A material like 4-4-2 applies N and elemental P in a ratio of 2.3:1. This ratio, when applied at rates to supply the N needs of spinach, provides over three times more P than is taken up by the crop. This relationship indicates that continued use of 4-4-2 could result in a buildup of P in the soil. However, our measurements showed that only 9% to 17% of the P content of 4-4-2 was released during the crop cycle. Whether the fertilizer was placed on the surface or buried in the soil did not affect this rate of release. It is not entirely clear why the release of soluble P from this material was low, but the two factors that may affect P release are the characteristics of the manure (Leytem and Mikkelsen 2005) and soil lime content (Hartz 2020).

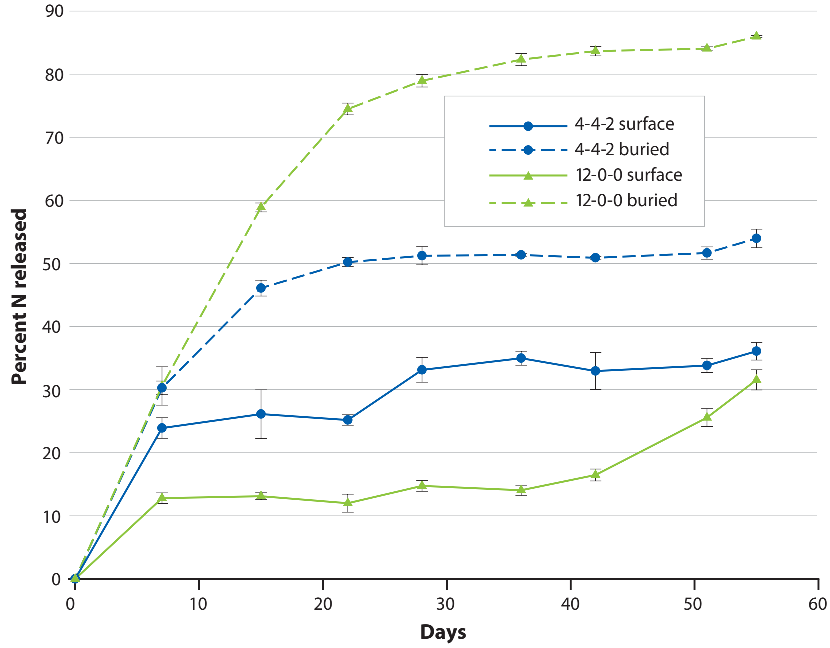


FIG. 2. Nitrogen released from pouches with 4-4-2 and 12-0-0 placed on the soil surface and buried three inches deep. Bars are standard error of the mean.

Tractor applying fertilizer. *Photo:* Richard Smith.

Pelleted organic fertilizer in a broccoli field. *Photo:* Richard Smith.

Stimulating microbial activity

Organic vegetable growers on the Central Coast face the same production pressures as conventional growers: high land rents, scheduling pressures and food safety concerns. These issues create barriers to using cover crops and compost, which generally were not used in the organic farms we evaluated. The biggest difference in the amount of carbon (C) added to the soil in organic versus conventional operations was the use of dry organic fertilizers. Materials like 4-4-2 and 12-0-0 contain 28% and 46% C, respectively. Additionally, 9,000 lb of 4-4-2 per year for three vegetable crops adds 2,520 lb/ac of C. For comparison, the 7,000 lb of above-ground dry matter from a cover crop contains 3,080 lb of C (Brennan and Smith 2017). Yearly applications of substantial amounts of 4-4-2 and other organic fertilizers provide significant inputs of C to the soil. We expected to observe greater levels of organic matter and total C in the organic farms than in the paired conventional farms. There was a great deal of variability in the levels of soil organic matter and total soil C in the two systems, and no statistical difference between them was detected (table 2). However, organic fields had significantly higher levels of WEON and FDA hydrolysis ($P < 0.05$) than did neighboring conventional fields. Higher levels of FDA hydrolysis in the organic systems indicates higher soil microbial activity (Green et al. 2006). These data indicate that large-scale organic vegetable operations that seldom use cover crops or compost can benefit from the extensive use of organic fertilizers to stimulate soil microbial activity.

In the 20 fields we evaluated, we observed that the net amount of N supplied by organic fertilizers applied to cool-season vegetables was generally less than crop uptake. This was because organic fertilizers release only a portion of the N they contain during the crop cycle. It is likely that the modest amounts of residual soil N and N in irrigation water made up the difference necessary for acceptable yields. We found that the time lag in release of N from these fertilizers and nitrate leaching on sandy soils adds uncertainty to making decisions for fast-maturing crops like spinach. We also found that the use of high P:N fertilizer, such as 4-4-2, did not significantly build up bicarbonate extractable P levels in Salinas Valley soils. Moving forward, testing for residual soil nitrate can help guide growers in their organic fertilizer decisions, especially for heavier soils. CA

R. Smith is Vegetable Crop Farm Advisor, UC Cooperative Extension (UCCE) Monterey, Santa Cruz and San Benito counties, Salinas; M. Cahn is Irrigation and Water Resources Farm Advisor, UCCE Monterey, Santa Cruz and San Benito counties, Salinas; T. Hartz is Extension Vegetable Specialist, Emeritus, Department of Plant Sciences, UC Davis; D. Geisseler is Extension Soils Specialist, Department of Land, Air and Water Resources, UC Davis; P. Love is Staff Research Assistant, UCCE Monterey County, Salinas.

We are grateful to the many growers who graciously cooperated with us on these studies and to the Fertilizer Research and Education Program of the California Department of Food and Agriculture that provided the funding for this study. We also thank Irfan Ainuddin for lab analyses. We appreciate insights and comments on this manuscript by Eric Brennan, USDA Research Horticulturalist.

References

Bottoms TG, Smith RF, Cahn MD, Hartz TK. 2012. Nitrogen requirements and N status determination of lettuce. HortScience 47(12):1768–74. https://doi.org/10.21273/HORTSCI.47.12.1768

Brennan EB, Smith RF. 2017. Cover crop frequency and compost effects on a legume-rye cover crop during eight years of organic vegetables. Agron J 109(5):2199–2213. https://doi.org/10.2134/agronj2016.06.0354

Breschini SJ, Hartz TK. 2002. Presidedress soil nitrate testing reduces nitrogen fertilizer use and nitrate leaching hazard in lettuce production. HortScience 36(7):1061–4. https://doi.org/10.21273/HORTSCI.37.7.1061

Cahn MD, Smith RF, Murphy L, Hartz TK. 2017. Field trials show the fertilizer value of nitrogen in irrigation water. Calif Agr 71(2):62–7. https://doi.org/10.3733/ca.2017a0010

Culman SW, Snapp SS, Freeman MA, et al. 2012. Permanganate oxidizable carbon reflects a processed soil fraction that is sensitive to management. Soil Sci Soc Am J 76:494–504. https://doi.org/10.2136/sssaj2011.0286

Gaskell MJ, Smith RF, Mitchell J, et al. 2000. Soil fertility management for organic crops. UC ANR Publication 7249. https://anrcatalog.ucanr.edu/Details.aspx?itemNo=7249

Gaskell MJ, Smith RF. 2007. Nitrogen sources for organic vegetable crops. HortTechnology 17(4):431–41. https://doi.org/10.21273/HORTTECH.17.4.431

Green VS, Stott DE, Diack M. 2006. Assay for fluorescein diacetate hydrolytic activity: Optimization for soil samples. Soil Biol Biochem 38:693–701. https://doi.org/10.1016/j.soilbio.2005.06.020

Haney RL, Franzluebbers AJ, Jin VL, et al. 2012. Soil organic C:N vs waterextractable organic C:N. Open J Soil Sci 2:269–74. http://doi.org/10.4236/ojss.2012.23032

Hartz TK, Bendixen WE, Weirdsma L. 2000. The value of presidedress soil nitrate testing as nitrogen management tool in irrigated vegetable production. HortScience 35(4):651–6. https://doi.org/10.21273/HORTSCI.35.4.651

Hartz TK, Johnstone PR. 2006. Nitrogen availability from high-nitrogen containing organic fertilizers. HortTechnology 16(1):39–42. https://doi.org/10.21273/HORTTECH.16.1.0039

Hartz TK. 2020. Efficient nutrient management in California vegetable production. Davis: UC Agriculture and Natural Resources Publication 3555. ISBN-13: 978-1-62711-070-9. https://anrcatalog.ucanr.edu/Details.aspx?itemNo=3555

Heinrich A, Smith RF, Cahn MD. 2013.Nutrient and water use of fresh market spinach. HortTechnology 23(3):325–33. https://doi.org/10.21273/HORTTECH.23.3.325

Johnstone PR, Hartz TK, Cahn MD, Johnstone MR. 2005. Lettuce response to phosphorus fertilization in high phosphorus soils. HortScience 40(5):1499–1505. https://doi.org/10.21273/HORTSCI.40.5.1499

Johnson, LF, Cahn M, Martin F, et al. 2016. Evapotranspiration-based irrigation scheduling of head lettuce and broccoli. HortScience 51(7):935–40. https://doi.org/10.21273/HORTSCI.51.7.935

Lazicki P, Geisseler D, Lloyd M. 2020. Nitrogen mineralization from organic amendments is variable but predictable. J Environ Qual 49(2):483–95. https://doi.org/10.1002/jeq2.20030

Leytem, AB, Mikkelsen RL. 2005. The nature of phosphorus in calcareous soils. Better Crops 89(2):11–3. https://eprints.nwisrl.ars.usda.gov/id/eprint/14/1/1159.pdf

Maltais-Landry E, Scow K, Brennan EB, et al. 2016. Higher flexibility in input N:P ratios results in more balanced phosphorus budgets in two long-term experimental agroecosystems. Agric Ecosyst Environ 223:197–210. https://doi.org/10.1016/j.agee.2016.03.007

Miller K, Aegerter BJ, Clark NE, et al. 2018. Relationship between soil properties and nitrogen mineralization in undisturbed soil cores from California agroecosystems. Comm Soil Sci Plant Anal 50:77–92. https://doi.org/10.1080/00103624.2018.1554668

Using Ecological Site Descriptions to make ranch-level decisions about where to manage for soil organic carbon

Rangeland conservation can keep carbon out of the atmosphere by storing it in the soil. Ecological Site Descriptions can help determine promising sites.

by Lina Aoyama, James W. Bartolome, Lucas Silva and Whendee L. Silver

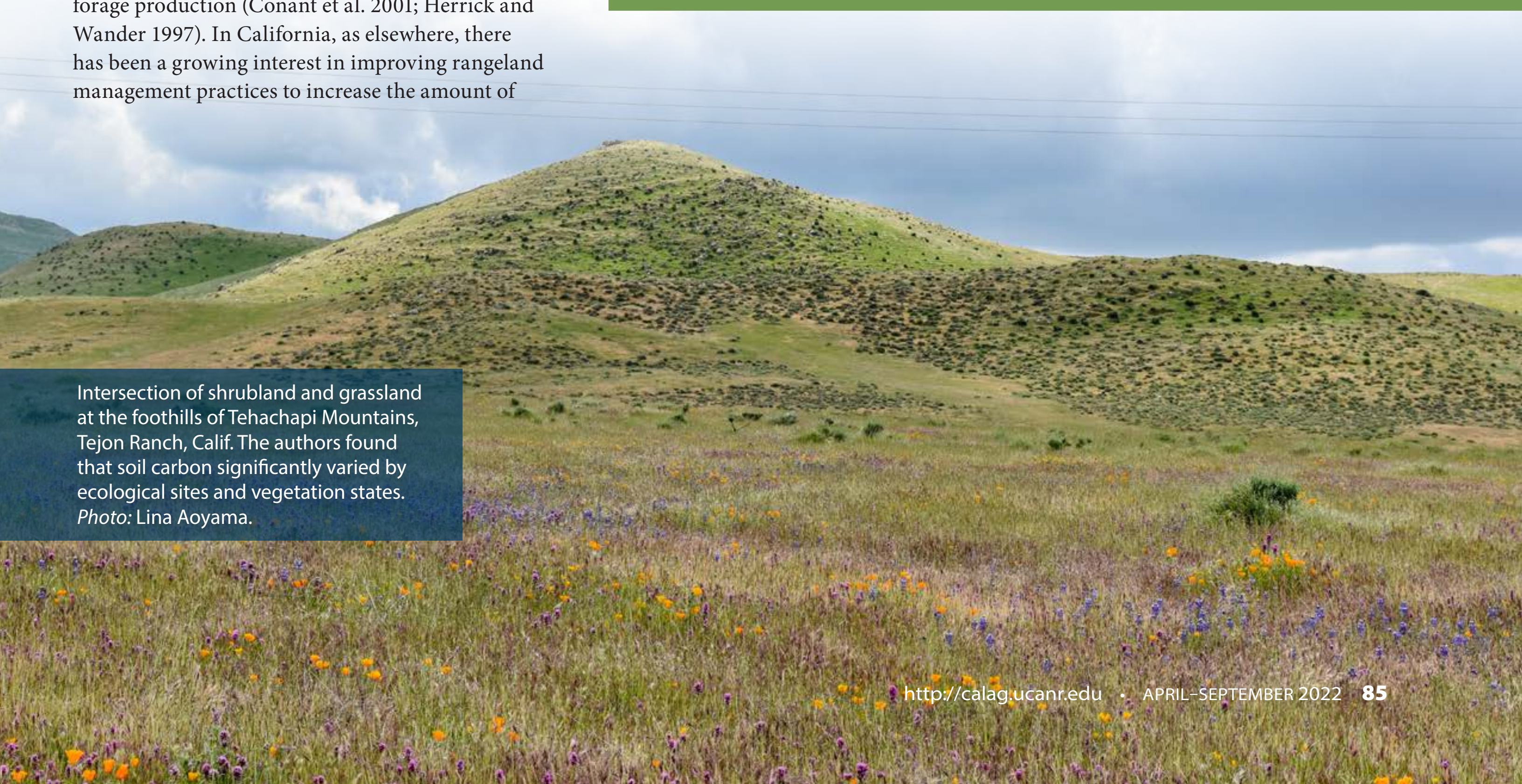
Online: https://doi.org/10.3733/ca.2022a0007

Abstract

Maintaining and enhancing soil organic carbon storage can mitigate climate change while promoting forage growth. California has adopted incentive programs to promote rangeland practices that build soil organic carbon. However, there is no standard framework for assessing the baseline level of soil organic carbon at the ranch scale. Here, we use the Ecological Site Description — a land-type classification system — to help ranch managers set priorities about where to implement practices to increase soil organic carbon. We measured baseline carbon stocks at 0 to 15 and 15 to 30 centimeters' depth across three ecological sites and two vegetation states (shrubland and grassland) at Tejon Ranch, California. We discovered increased levels of soil carbon at ecological sites in higher elevations, and more soil carbon in shrublands as compared to grasslands. Slope, elevation, and soil texture, as well as plant litter and shrub cover, were significant predictors of soil carbon. The Ecological Site Description framework can serve as an important tool to help range managers keep carbon in the soil and out of the atmosphere.

California's rangelands cover approximately 57 million acres of grasslands, shrublands, woodlands and deserts (FRAP 2018). As the most extensive land use type in California, rangelands play an important role in climate change mitigation by storing considerable amounts of soil organic carbon (SOC) below ground (Herrero et al. 2016; Sanderson et al. 2020; Schuman et al. 2002). In wildfire and drought-prone California, rangelands are considered more reliable for carbon storage because they secure most of the carbon in the soil instead of in aboveground plants that could potentially burn (Dass et al. 2018).

The primary way that carbon is stored in the soil is as soil organic matter (SOM). Soil organic matter is a mix of decomposing plant and animal biomass, soil microbes and humus. Increasing SOM also enhances soil water-holding capacity and improves nutrient cycling, which can maintain or increase forage production (Conant et al. 2001; Herrick and Wander 1997). In California, as elsewhere, there has been a growing interest in improving rangeland management practices to increase the amount of



Intersection of shrubland and grassland at the foothills of Tehachapi Mountains, Tejon Ranch, Calif. The authors found that soil carbon significantly varied by ecological sites and vegetation states. *Photo:* Lina Aoyama.

SOC (Bradford et al. 2019; Byrnes et al. 2017; Carey, Gravuer et al. 2020).

The California Department of Food and Agriculture (CDFA)'s Healthy Soils Program is one of the programs that has been put in place to promote carbon storage. This program provides financial incentives through cost sharing and technical support for land managers to implement agricultural management practices that augment SOC or prevent erosion (CDFA 2016). Between 2017 and 2020, 604 landowners participated in this program, and the estimated greenhouse gas reduction was 109,089 tonnes CO_2 eq/year (CDFA 2021). Recommended range management practices include planting oak trees, applying compost, and restoring riverbanks (Dahlgren et al. 1997; Matzek et al. 2018; Ryals et al. 2014).

Ecological Site Descriptions are a useful framework for assessing baseline soil organic carbon on diverse landscapes.

Ideally, land managers should be able to easily find out the baseline amount of soil organic carbons on their lands and evaluate where to implement these practices. However, the amount of SOC varies greatly from place to place because of California's diverse climate, topography and soil conditions (Carey, Weverka et al. 2020). The Rangeland Monitoring Network is currently evaluating the variability in soil carbon storage potential across California rangelands (Carey et al. 2020b). There are SOC estimates by soil classification available on the USDA Web Soil Survey, but we lack a framework to relate these soil carbon estimates to specific landscape features at a ranch scale.

Ecological Site Description (ESD), developed and maintained by the USDA Natural Resource Conservation Service, is a land-type classification system that could facilitate ranch-level planning to manage for soil carbon. The ESD framework has primarily been used to manage vegetation on rangelands and forests, as well as to identify priorities for ecosystem benefits such as wildlife habitat, water quality and wildfire protection (Brown and Havstad 2016). This framework has not yet been extended to manage SOC stocks. An ecological site is defined as an area 24.7 to 247 acres with similar "physical properties [climate, soils, topography, underlying geologic material], potential vegetation, and responses to management that differ from other kinds of land" (USDA 2018). Each ecological site contains a state-and-transition (STM) model of vegetation states that is helpful in evaluating the effects of management actions or disturbances on the existing state of vegetation (Briske et al. 2005).

A ranch can have more than one ecological site. A related framework called Major Land Resource Area (MLRA) has been used by the National Resource Inventory Soil Monitoring Network to estimate SOC stocks at a national scale (Ogle et al. 2010; Spencer et al. 2012). However, these estimates are not precise enough for ranch-level decision making. While similar in concept, ESD units are smaller, more site-specific than MLRAs and may be more fitting for ranch-level planning.

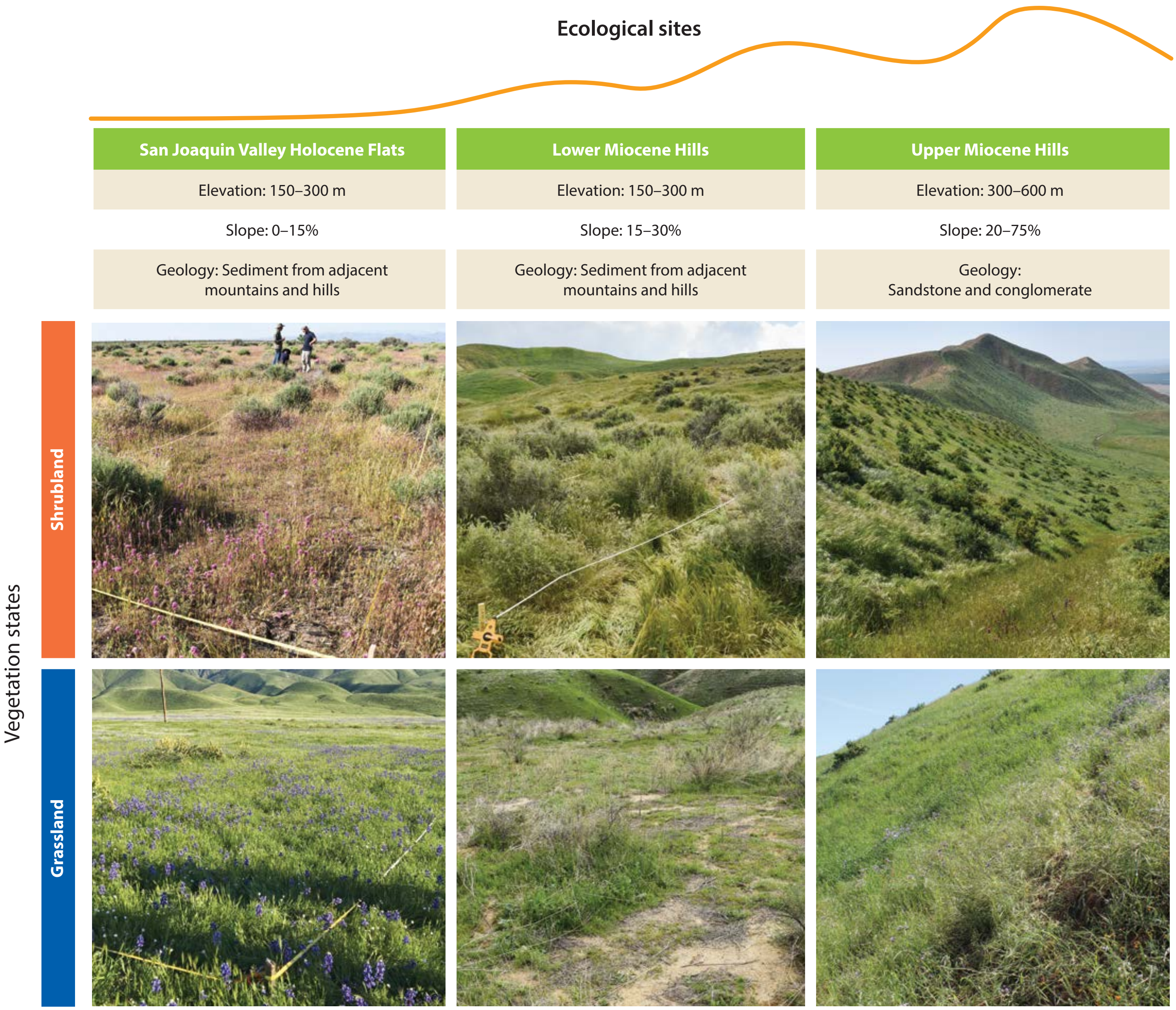
The goal of this study is to explore the use of ESDs for ranch-level identification of potential priority sites for soil carbon sequestration projects. We used Tejon Ranch in Kern County, California, as a case study. Tejon Ranch is an ideal place to study ESDs because it has complex biogeographical features, and its range managers already use ESDs to manage forage for livestock (Spiegal et al. 2016). Studies connecting soil carbon to plant communities are scarce in Southern California (Booker et al. 2013; Carey et al. 2020b). To examine how soil carbon content is related to ESDs and vegetation states, we measured topsoil carbon in three ecological sites at different elevations and in two dominant vegetation states. We then explored how plant and soil characteristics were related to patterns in SOC stocks at ranch level.

Ecological sites on Tejon Ranch

Tejon Ranch (270,000 acres) is the largest contiguous, privately owned property in California, located 35 miles south of Bakersfield, California (latitude 34.935044°, longitude 118.670405°). We studied the northwestern portion of the ranch (50,000 acres) where the San Joaquin Valley and Tehachapi Mountains meet. Using the ESD framework, three ecological sites were defined based on slope, elevation and geology (fig. 1): Holocene Flats, Lower Miocene Hills and Upper Miocene Hills. Tejon Ranch has been grazed by livestock since the 1840s when the original Tejon Ranchos were created (Latta 1976). During the study, beef cattle (*Bos taurus*) grazed year-round while no grazers, including wildlife, were excluded from the study area.

Dominant vegetation types in the study site were native shrubland and non-native annual grassland. Native shrub species commonly found in shrublands included cattle saltbrush (*Atriplex polycarpa*), Interior California buckwheat (*Eriogonum fasciculatum var. polifolium*) and Valley bladderpod (*Peritoma arborea var. globose*) (Aoyama et al. 2020). The grassland state was primarily composed of native and non-native annual forbs (e.g., *Plagiobothrys canescens*, *Erodium cicutarium*) and non-native annual grasses (e.g., *Bromus diandrus*, *Avena barbata*) (Spiegal et al. 2016).

Soils were generally shallow Mollisols with bedrock encountered at approximately 30 centimeters' (cm) depth. Parent materials were sedimentary rocks from the Miocene and alluvium deposits from the Holocene (Dibblee 2008). Soil classification varied from Psamments-Xerolls complex (loamy sand), to



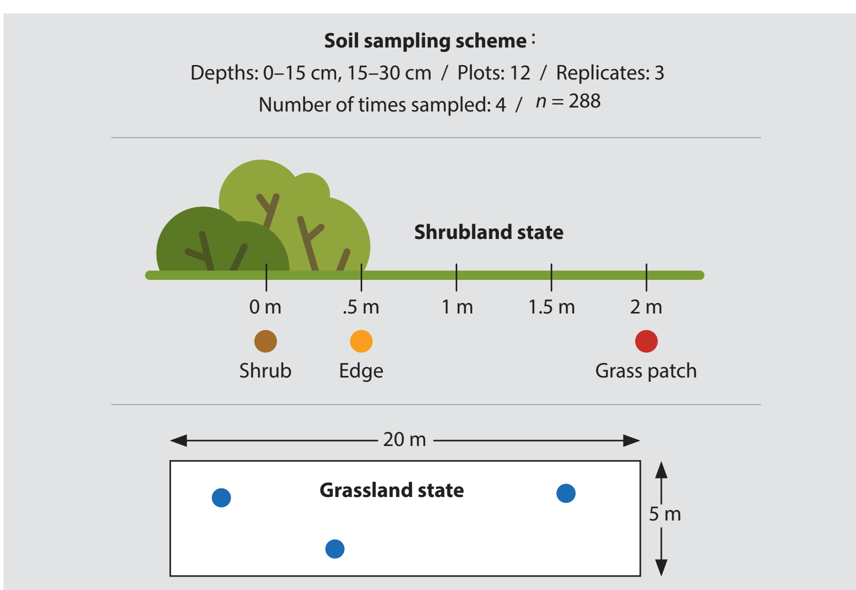


FIG. 1. Ecological sites along the elevational gradient of Tehachapi Mountain in Tejon Ranch, California: San Joaquin Valley Holocene Flats, Lower Miocene Hills and Upper Miocene Hills. The schematic is adapted from Spiegel et al. (2016). Each ecological site has two vegetation types: shrubland (top) and grassland (bottom). Two plots in each vegetation type are nested within each ecological site. Each plot was sampled for soil, cattle use and vegetation cover. Three replicates of soil cores were collected at each plot four times. Soils in shrubland plots were collected from three microhabitats: base of shrub, edge of shrub canopy (~0.5 m from the base) and grass patch between shrubs (~2 m from the base). Soils in grassland plots were collected randomly.

TABLE 1. Heterogeneous soil properties across the landscape

Ecological Site	pH	PO_4^{3-}	K^+	Na^+	Ca^{2+}	Mg^{2+}	SO_4^{2-}	Sand	Clay
	[H+]	*ppm*	*Meq/100 g*	*Meq/100 g*	*Meq/100 g*	*Meq/100 g*	*ppm*	*%*	*%*
SJV Holocene Flats	7.56	5.24	0.66	0.05	7.96	1.18	3.53	83.23	7.24
	(0.70)	(2.15)	(0.51)	(0.06)	(5.69)	(0.82)	(3.33)	(8.70)	(3.86)
Lower Miocene Hills	7.64	4.45	0.58	1.53	11.38	1.38	8.25	74.47	9.80
	(0.32)	(1.48)	(0.69)	(2.99)	(7.71)	(0.39)	(12.12)	(13.26)	(4.97)
Upper Miocene Hills	7.32	3.52	0.62	0.24	23.57	3.50	5.57	60.82	20.86
	(0.67)	(1.95)	(0.15)	(0.10)	(5.52)	(1.58)	(2.41)	(5.14)	(7.82)
ANOVA (*P*-value)	0.70	0.71	0.78	0.48	**0.04**	**0.04**	0.72	0.08	**0.03**

Values are mean and standard error of each soil property by ecological sites.
The last row contains *P*-values from ANOVAs of soil properties with ecological site as a main effect. Significant *P*-values ($P \leq 0.05$) are bolded.

Haploxerolls (sandy loam), to Chanac-Pleito complex (sandy clay loam) (USDA Web Soil Survey); the clay content increased with elevation (table 1).

We established twelve plots (65 feet by 16 feet) stratified by three ecological sites and two dominant vegetation types (grassland versus shrubland; fig. 1). We measured plant cover (shrub, herbaceous, litter, bare and other) and cattle use (evidence of herbivory, trailing, trampling, and old and new cattle manure) in each plot in March, June and October 2017 and March 2018. In addition to qualitative measures of cattle use, we set up motion-triggered camera traps on t-posts at heights of 1.5 meters on each plot to measure the number of cattle visits from March 2017 to March 2018.

To measure soil carbon concentrations and content, we sampled three replicate cores in each plot from two depths (0–15 cm and 15–30 cm) using a 5-cm diameter by 15-cm-long soil corer. We sampled the topsoil because it is the fraction of soil that is most accessible, where management has the most impact on soil carbon on an annual basis (Fontaine et al. 2007; Gregory et al. 2016; Syswerda et al. 2011). We sampled soils within grassland plots randomly; within shrubland plots, we stratified by microhabitats at the base of shrub main stem, at the edge of shrub canopy, and at a grass patch 2 meters from the shrub main stem. Cores were weighed fresh and after drying at 105°C for bulk density determination (see online technical appendix). We measured the concentration of SOM using the loss-on-ignition method and the concentration of SOC using the flash combustion method. We then converted concentration percentages to stocks (Mg/ha) by multiplying the values by bulk density and depth. SOC was positively correlated with SOM [log(SOC Mg/ha) = 0.68 * log(SOM Mg/ha) + 0.18; $R^2 = 0.41$, $P < 0.001$].

Mary McDonnell, Dylan Stover and Tara Harmon (from right to left) taking turns to collect soil in a grassland plot, Tejon Ranch, Calif. *Photo:* Lina Aoyama.

Soil carbon by vegetation and ecological site

Our study explored the idea of using ESDs to provide a first approximation of baseline soil carbon at ranch level. In the study area on Tejon Ranch, average SOC stocks at 0–15 cm, 15–30 cm and 0–30 cm depths were 19.18 ± 0.90 Mg C/ha, 14.25 ± 0.67 Mg C/ha and 33.63 ± 1.49 Mg C/ha, respectively (table 2). These estimates were less than the statewide average (58 ± 4 Mg C/ha in 0–25 cm) reported by Silver et al. (2010). In contrast, these values are comparable to the statewide average (19.31 Mg C/ha in 0–10 cm) reported by Carey et al. (2020) and the 20.98 Mg C/ha for the A horizon in the Sierra Nevada foothills (Eastburn et al. 2017). From the linear mixed-effect models, we found that SOM and SOC significantly varied by ecological sites and vegetation states (table 3).

Both SOM and SOC stocks increased with elevation. They were higher in the Upper Miocene Hills than in the San Joaquin Valley Holocene Flats ecological sites (table 2). This effect of elevation could be due to a combination of time, soil type, and legacy of historical plant communities. Soils in the Miocene Hills were older than those in the Holocene Flats

TABLE 2. Bulk density (g/cm^3), soil organic matter (SOM) concentration (%), soil organic carbon (SOC) concentration (%), SOM content (Mg/ha) and SOC content (Mg/ha) by depth and spatial scale

Depth	Scale		Bulk density (g/cm^3)	SOM concentration (%)	SOC concentration (%)	SOM content (Mg/ha)	SOC content (Mg/ha)
0–15 cm	Landscape		1.11 ± 0.01	3.13 ± 0.12	1.18 ± 0.05	49.09 ± 1.64	19.18 ± 0.90
	ES	SJV_Holo	1.17 ± 0.01	2.24 ± 0.13	0.91 ± 0.06	39.38 ± 2.19	15.81 ± 0.77
		Low_Mio	1.20 ± 0.01	2.75 ± 0.22	1.06 ± 1.13	46.73 ± 3.30	19.64 ± 2.42
		Up_Mio	0.94 ± 0.01	4.42 ± 0.13	1.58 ± 0.06	61.68 ± 2.12	22.26 ± 0.93
	Veg	Grassland	1.17 ± 0.01	2.69 ± 0.16	0.89 ± 0.05	44.49 ± 1.85	14.87 ± 0.67
		Shrubland	1.05 ± 0.01	3.54 ± 0.17	1.45 ± 0.08	53.49 ± 2.60	23.29 ± 1.53
15–30 cm	Landscape		1.13 ± 0.01	0.84 ± 0.03	0.84 ± 0.03	37.84 ± 1.17	14.25 ± 0.67
	ES	SJV_Holo	1.20 ± 0.01	1.77 ± 0.08	0.73 ± 0.04	33.08 ± 1.39	12.87 ± 0.65
		Low_Mio	1.23 ± 0.01	1.92 ± 0.14	0.79 ± 0.08	34.08 ± 2.26	14.24 ± 1.78
		Up_Mio	0.95 ± 0.01	3.44 ± 0.13	1.03 ± 0.04	47.77 ± 1.84	15.81 ± 0.60
	Veg	Grassland	1.18 ± 0.01	2.13 ± 0.15	0.60 ± 0.03	35.95 ± 1.83	10.42 ± 0.46
		Shrubland	1.09 ± 0.01	2.51 ± 0.10	1.05 ± 0.05	39.52 ± 1.50	17.64 ± 1.07
0–30 cm	Landscape					87.44 ± 2.49	33.63 ± 1.49
	ES	SJV_Holo				72.46 ± 2.64	28.68 ± 1.21
		Low_Mio				81.78 ± 5.01	35.02 ± 4.02
		Up_Mio				110.0 ± 3.39	37.66 ± 1.39
	Veg	Grassland				80.91 ± 3.34	25.48 ± 0.99
		Shrubland				93.46 ± 3.57	41.14 ± 2.47

Values are mean and standard error of each estimate. ES = ecological site, Veg = vegetation state.

TABLE 3. Ecological sites and vegetation types influenced bulk density, soil organic matter (SOM) and soil organic carbon (SOC)

		Ecological site		Vegetation state		Ecological site: vegetation state	
Response	*Depth*	*F*	*P-value*	*F*	*P-value*	*F*	*P-value*
Bulk density	0–15 cm	8.576	**0.017**	4.399	0.080	0.091	0.914
	15–30 cm	6.367	**0.032**	1.351	0.289	0.023	0.976
SOM (Mg/ha)	0–15 cm	3.543	**0.009**	1.565	0.257	2.691	0.146
	15–30 cm	15.043	**0.004**	1.194	0.316	14.283	**0.005**
	0–30 cm	6.357	**0.033**	1.069	0.341	4.680	0.059
SOC (Mg/ha)	0–15 cm	1.324	**0.033**	6.386	**0.044**	1.949	0.222
	15–30 cm	0.541	0.607	13.154	**0.011**	1.465	0.303
	0–30 cm	1.744	0.252	10.393	**0.018**	1.8025	0.243

F statistics and *P*-values are results of linear mixed-effect models of SOM and SOC with ecological site, vegetation state, and their interactions as main effects and plot as a random effect. Significant *P*-values ($P \leq 0.05$) are bolded.

(Dibblee 2008), and thus had more time to accumulate soil carbon. High clay content found in higher elevation sites also may have contributed to higher soil carbon storage due to carbon-mineral associations (Kaiser et al. 2012). Additionally, oak savanna historically dominated the slopes of the Tehachapi Mountains (Twisselman 1967), which likely contributed to higher soil carbon inputs on the slopes than on the flats (Dahlgren et al. 1997; Koteen et al. 2011).

We also found that both SOM and SOC stocks were significantly higher in shrubland states compared to grassland states in the 0–30 cm profile (table 2), which corroborates the broad pattern found in California's rangelands that woody plants increase soil carbon pools (Silver et al. 2010). One possible explanation for this is that shrubs in semi-arid environments tend to have dense root mats in the top 30 cm to capture transient soil moisture (Chabbi et al. 2009; Swanston et al. 2005), which contribute to soil carbon storage. Another possible explanation is that litter fall from shrubs contributes significantly to near-surface soil carbon pools (Rau et al. 2009). These results suggest that grouping the vegetation states by dominant functional groups within

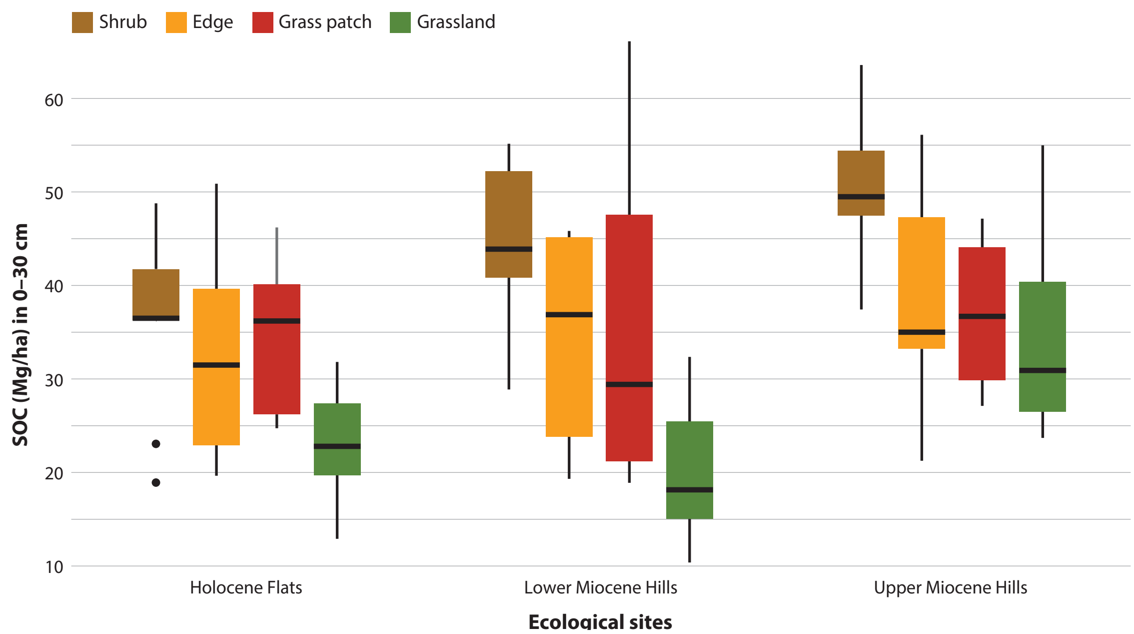


FIG. 2. Soil organic carbon (SOC) stocks in 0–30 cm depth by ecological sites and microhabitats: shrub, edge of shrub canopy, grass patch between shrubs, and grassland. Horizontal bars are medians, boxes are the 25th and 75th percentile, and the whiskers are the maximum and minimum values. Dots represent outliers in the data.

ecological sites could be valuable for evaluating SOC stocks at ranch level.

Soil carbon varied across microhabitats

Given the wide range of sizes of ranches in California, the scalability of plot-level information is important. The spatial variability of SOC stocks within a ranch informs managers about the spatial scale at which they should aggregate the estimates of SOC from the USDA Web Soil Survey or their own soil sampling. The average coefficient of variance of SOC stocks in shrubland was higher (85.9) than that in grassland (38.67). This plot-level variability came from the microhabitat types (shrub base, edge of shrub canopy, grass patch between shrubs, and grassland). Estimates of SOC stocks in 0–30 cm were significantly higher at the base of shrubs than in grasslands ($F_{3,119} = 7.924$, $P < 0.001$; fig. 2). These results support the notion of shrubs acting as "islands of fertility" where soil carbon and nutrients are high relative to areas outside the canopy (Schlesinger et al. 1996; Schade et al. 2003; Schade and Hobbie 2005). The implication for SOC monitoring is that sampling more cores in shrubland than in grassland is necessary to capture the spatial variability in soil carbon within vegetation states.

Predictors of carbon across the landscape

Grazing management has been proposed as a means to sequester soil carbon (Byrnes et al. 2017). However, in the literature, cattle use is often not well quantified beyond presence-absence data. We used the standard qualitative assessment of cattle use (e.g., evidence of herbivory, fresh and old cow manure, trampling) and quantified the frequency and intensity of cattle use via camera traps. We found that none of these grazing use metrics meaningfully captured variation in SOM or SOC stocks. This result is in line with prior work showing that the influence of grazing on soil carbon is not significant in semi-arid rangelands of California (Biggs and Huntsinger 2021), especially in sandier soils (Silver et al. 2010; Stanton et al. 2018).

To identify important factors that predict soil carbon at ranch level, we examined the relationships between environmental variables and soil carbon stocks. We found that mean percent cover of litter and shrubs predicted SOM stocks in the 0–30 cm profile, while elevation, slope and soil calcium content predicted SOC stocks in the 0–30 cm profile (table 4). These were similar to the factors that influenced soil carbon at the regional level (Carey et al. 2020). Managers may use this information to group their ranch or management unit by ecological sites and vegetation states, and make decisions about where on the landscape they might want to implement carbon sequestration projects. For example, managers on Tejon Ranch could use the ESDs to prioritize conserving or restoring native shrubs in the high elevation Upper Miocene Hills ecological site. On other ranches, land managers might increase soil carbon by planting oak trees in ecological sites that have oak woodland as a reference state, or by applying compost in ecological sites with low baseline

TABLE 4. Abiotic and biotic environmental factors influence soil organic matter (SOM) and soil organic carbon (SOC)

Response	Depth	Predictor	Coefficient	*t*-value	*P*-value	R^2_m	R^2_c
SOM	0–15 cm	Mg	0.458	1.732	0.121	0.489	0.963
		Litter % cover	0.514	1.675	0.132		
		Shrub % cover	0.454	1.803	0.109		
	15–30 cm	pH	0.244	1.442	0.208	0.602	0.964
		Ca	−0.399	−1.865	0.121		
		Clay	0.276	1.754	0.139		
		Slope	0.273	2.392	0.062		
		Litter % cover	0.732	3.843	**0.012**		
		Shrub % cover	0.421	3.217	**0.023**		
SOC	0–15 cm	Ca	0.622	0.268	0.059	0.419	0.945
		Clay	−0.557	0.304	0.116		
		Elevation	−0.989	0.361	**0.034**		
		Slope	0.953	0.253	**0.009**		
		Litter % cover	0.637	0.300	0.077		
	15–30 cm	Ca	0.528	3.306	**0.013**	0.396	0.990
		Clay	−0.431	−2.596	**0.035**		
		Elevation	−0.375	−2.084	0.075		
		Slope	0.501	3.375	**0.011**		

Coefficients, *t*-values, *P*-values, R^2_m and R^2_c are results of linear regression models with SOM and SOC as response variables, environmental factors as fixed effects, and quadrats within plots as random effects. R^2_m is a proportion of variance explained by fixed effects and R^2_c is that explained by both fixed and random effects. Significant *P*-values ($P \leq 0.05$) are bolded.

SOC stocks. This approach is compatible with range management in California that values multiple ecosystem services supported by the landscape rather than single-purpose management (Biggs and Huntsinger 2021).

Deciding where to build up carbon

California is leading the way in building up carbon in the soil, where it helps hold nutrients and water, and out of the atmosphere, where it contributes to global warming. Planting oak trees on range land, adding compost, and maintaining rivers can increase carbon sequestration while improving forage. A simple framework to assess soil organic carbon stocks would make it easier for land managers to determine where to prioritize implementing practices that increase the amount of carbon stored in the soil. In this case study at Tejon Ranch, we demonstrated that Ecological Site Descriptions are a useful framework for assessing baseline soil organic carbon on diverse landscapes. We found that the amount of soil organic carbon differs by dominant vegetation functional groups and ecological sites. Environmental factors such as soil type, topography and vegetation cover are predictors of the amount of soil carbon.

ESDs are less developed in California than in other states. However, this framework can contribute to our understanding of the environmental factors that influence soil organic carbon. This should enable rangeland managers to focus soil management practices where they will do the most good. CA

L. Aoyama is Ph.D. student, Environmental Studies Program and Department of Biology, University of Oregon, Eugene, Oregon; J.W. Bartolome is Emeritus Professor, and W.L. Silver is Professor, Department of Environmental Science, Policy, and Management, UC Berkeley; L. Silva is Associate Professor, Environmental Studies Program, Department of Geography, and Institute of Ecology and Evolution, University of Oregon, Eugene, Oregon.

We thank Peter Hopkinson, Felix Ratcliff, Ellery Mayence, Atalie Brown, Andrew Capistrano, Jack Cronin, Denise Defreese, Lee Fram, Luke Macaulay, Mary McDonnell, Tara Harmon, Michael Orellana, Kea Rutherford, Allison Rofe, Dylan Stover and Lulu Waks for helping with fieldwork. We thank Summer Ahmed for assisting with soil analysis. Financial support was provided by the Tejon Ranch Conservancy, Range Block Grant, Russell Rustici Chair in Range Management, project numbers CA-B-ECO-0117-MS and CA-B-ECO-7673-MS of the USDA National Institute of Food and Agriculture, the University of California, Berkeley, and The Nature Conservancy under the Oren Pollak Memorial Student Research Grant for Grassland Science.

References

Aoyama L, Bartolome JW, Hallett LM. 2020. Incorporating diversity measures into Ecological Site Descriptions to manage biodiversity on heterogeneous landscapes. Rangelands 42(4):93–105. https://doi.org/10.1016/j.rala.2020.05.002

Biggs NB, Huntsinger L. 2021. Managed grazing on California annual rangelands in the context of state climate policy. Rangeland Ecol Manag 76:56–68. https://doi.org/10.1016/j.rama.2021.01.007

Booker K, Huntsinger L, Bartolome JW, et al. 2013. What can ecological science tell us about opportunities for carbon sequestration on arid rangelands in the United States? Global Environ Chang 23:240–51. https://doi.org/10.1016/j.gloenvcha.2012.10.001

Bradford MA, Carey CJ, Atwood L, et al. 2019. Soil carbon science for policy and practice. Nat Sustain 2:1070–2. https://doi.org/10.1038/s41893-019-0431-y

Briske DD, Fuhlendorf SD, Smeins FE. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecol Manag 58:1–10. www.jstor.org/stable/3899791

Brown JR, Havstad KM. 2016. Using ecological site information to improve landscape management for ecosystem services. Rangelands 38(6):318–21. https://doi.org/10.1016/j.rala.2016.10.011

Byrnes R, Eviner V, Kebreab E, et al. 2017. Review of research to inform California's climate scoping plan: Agriculture and working lands. Calif Agr 71:160–8. https://doi.org/10.3733/ca.2017a0031

Carey C, Gravuer K, Gennet S, et al. 2020. Supporting evidence varies for rangeland management practices that seek to improve soil properties and forage production in California. Calif Agr 74(2):101–11. https://doi.org/10.3733/ca.2020a0015.

Carey C, Weverka J, DiGaudio R, et al. 2020. Exploring variability in rangeland soil organic carbon stocks across California (USA) using a voluntary monitoring network. Geoderma Regional 22:e00304. https://doi.org/10.1016/j.geodrs.2020.e00304

[CDFA] California Department of Food and Agriculture. 2016. Healthy Soil Program Incentives Program. www.cdfa.ca.gov/oefi/healthysoils/IncentivesProgram.html (accessed Mar. 1, 2019).

CDFA. 2021. Healthy Soil Program Incentives Program. HSP Incentives Program Program-Level Data of Executed and Funded Grant Projects. www.cdfa.ca.gov/oefi/healthysoils/docs/HSP_Incentives_program_level_data_funded_projects.pdf (accessed Dec. 29, 2021).

Chabbi A, Kogel-Knabner I, Rumpel C. 2009. Stabilized carbon in subsoil horizon is located in spatially distinct parts of the soil profile. Soil Biol Biochem 41:256–61. https://doi.org/10.1016/j.soilbio.2008.10.033

Conant RT, Paustain K, Elliott E. 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecol Appl 11:343–55. https://doi.org/10.1890/1051-0761(2001)011[0343:GMACIG]2.0.CO;2

Dahlgren RA, Singer MJ, Huang X. 1997. Oak tree and grazing impacts on soil properties and nutrients in California oak woodland. Biogeochemistry 39:45–64. https://doi.org/10.1023/A:1005812621312

Dass P, Houlton BZ, Wang Y, Warlind D. 2018. Grasslands may be more reliable carbon sinks than forests in California. Environ Res Lett 13:074027. https://doi.org/10.1088/1748-9326/aacb39

Dibblee TW. 2008. Geologic map of the Cummings Mountain and Tehachapi quadrangles. Dibblee Geology Center Map #DF-397. Santa Barbara Museum of Natural History.

Eastburn DJ, O'Geen AT, Tate KW, Roche LM. 2017. Multiple ecosystem services in a working landscape. PLoS ONE 12:e0166565. https://doi.org/10.1371/journal.pone.0166595

[FRAP] CAL FIRE Fire and Resource Assessment Program. 2018. California's Forests and Rangelands 2017 Assessment. https://frap.fire.ca.gov/assessment/ (accessed Mar. 1, 2019).

Gregory AS, Dungait JAJ, Watts CW, et al. 2016. Long-term management changes topsoil and subsoil organic carbon and nitrogen dynamics in a temperate agricultural system. Eur J Soil Sci 67(4):421–30. https://doi.org/10.1111/ejss.12359

Herrero M, Henderson B, Havlik P, et al. 2016. Greenhouse gas mitigation potentials in the livestock sector. Nat Clim Change 6:452–61. https://doi.org/10.1038/nclimate2925

Herrick JE, Wander MM. 1997. Relationships between soil organic carbon and soil quality in cropped and rangeland soils: the importance of distribution, composition, and soil biological activity. In *Soil Processes and the Carbon Cycle*. Lal R, Kimble JM, Follett RF, Stewart BA (eds). New York, NY: CRC Press LLC. p 405–25.

Kaiser M, Ellerbrock RH, Wulf M, et al. 2012. The influence of mineral characteristics on organic matter content, composition, and stability of topsoils under long-term arable and forest land use. J Geophys Res 117:G02018. https://doi.org/10.1029/2011JG001712

Koteen LE, Baldocchi DD, Harte J. 2011. Invasion of non-native grasses causes a drop in soil carbon storage in California grasslands. Environ Res Lett 6:044001. https://doi.org/10.1088/1748-9326/6/4/044001

Latta FF. 1976. *Saga of Rancho El Tejon*. Exeter, Calif.: Bear State Books. 293 p.

Matzek V, Stella J, Ropion P. 2018. Development of a carbon calculator tool for riparian forest restoration. Appl Veg Sci 21:584–94. https://doi.org/10.1111/avsc.12400

Ogle SM, Breidt FJ, Easter M, et al. 2010. Scale and uncertainty in modeled soil organic carbon stock changes for US croplands using a process-based model. Glob Change Biol 16(2):810–22. https://doi.org/10.1111/j.1365-2486.2009.01951.x

Rau BM, Johnson DW, Blank RR, Chambers JC. 2009. Soil carbon and nitrogen in a Great Basin pinyon-juniper woodland: influence of vegetation, burning, and time. J Arid Environ 73(4-5):472–9. http://doi.org/.10.1016/j.jaridenv.2008.12.013

Robertson GP, Sollins P, Ellis BG, Lajtha K. 1999. Exchangeable ions, pH, and cation exchange capacity. In *Standard Soil Methods for Long-Term Ecological Research*. Robertson GP, Coleman DC, Blendsoe CS, Sollins P (eds.). New York, New York: Oxford Press. p 106–14.

Ryals R, Kaiser M, Torn MS, et al. 2014. Impacts of organic matter amendments on carbon and nitrogen dynamics in grassland soils. Soil Biol Biochem 68:52–61. https://doi.org/10.1016/j.soilbio.2013.09.011

Sanderson JS, Beutler C, Brown JR, et al. 2020. Cattle, conservation, and carbon in the western Great Plains. J Soil Water Conserv 75:5A–12A. https://doi.org/10.2489/jswc.75.1.5A

Schade JD, Sponseller R, Collins SL, Stiles A. 2003. The influence of Prosopis canopies on understorey vegetation: Effects of landscape position. J Veg Sci 14(5):743–50. https://doi.org/10.1111/j.1654-1103.2003.tb02206.x

Schade JD, Hobbie S. 2005. Spatial and temporal variation in islands of fertility in the Sonoran Desert. Biogeochemistry 73:541–53. https://doi.org/10.1007/s10533-004-1718-1

Schlesinger WH, Raikes JA, Hartley AE, Cross AF. 1996. On the spatial pattern of soil nutrients in desert ecosystems. Ecology 77(2):364–74. www.resolutionmineeis.us/documents/schlesinger-et-al-1996

Schuman GE, Janzen HH, Herrick JE. 2002. Soil carbon dynamics and potential carbon sequestration by rangelands. Environ Pollut 116:391–6. http://doi.org/S0269-7491(01)00215-9

Silver WL, Ryals R, Eviner V. 2010. Soil carbon pools in California's annual grassland ecosystems. Rangeland Ecol Manag 63:128–36. http://doi.org/10.2111/REM-D-09-00106.1

Spencer S, Ogle SM, Breidt FJ, et al. 2012. Designing a national soil carbon monitoring network to support climate change policy: a case example for US agricultural lands. Greenhouse Gas Measurement Manage 1(3-4):167–78. https://doi.org/10.1080/20430779.2011.637696

Spiegal S, Bartolome JW, White M. 2016. Applying ecological site concepts to adaptive conservation management on an iconic Californian landscape. Rangelands 38(6):365–70. https://doi.org/10.1016/j.rala.2016.10.002

Stanton CY, Mach KJ, Turner PA, et al. 2018. Managing cropland and rangeland for climate mitigation: an expert elicitation on soil carbon in California. Climatic Change 147:633–46. https://doi.org/10.1007/s10584-018-2142-1

Swanston CW, Torn MS, Hanson PJ, et al. 2005. Initial characterization of processes of soil carbon stabilization using forest stand-level radiocarbon enrichment. Geoderma 128(1-2):52–62. www.osti.gov/servlets/purl/15016762.

Syswerda SP, Corbin AT, Mokma DL, et al. 2011. Agricultural management and soil carbon storage in surface vs. deep layers. Soil Sci Soc Am J 75(1):92–101. https://doi.org/10.2136/sssaj2009.0414

Twisselman EC. 1967. A flora of Kern County, California. Wasmann J Biol 25:1–395.

[USDA] US Department of Agriculture. 2018. Ecological Site Descriptions. www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/ (accessed March 1, 2019).

RESEARCH ARTICLE

Improvements to the soil nitrate quick test for California small grains

Inexpensive soil nitrate quick tests can help small grain growers identify their crops' nitrogen fertilizer needs.

by Taylor S. Nelsen, Michael Rodriguez, Ethan McCullough, Serena N. Lewin, Daniel Geisseler, Konrad Mathesius, Taylor Becker, Gabriel G. Rosa and Mark E. Lundy

Online: https://doi.org/10.3733/ca.2022a0009

Nitrogen (N) fertilizer is important for crop production, but applying excessive amounts wastes resources and has negative environmental consequences. Many winter small-grain crops are rainfed and grown in areas with unpredictable precipitation patterns. Excess precipitation can cause nitrate to leach into groundwater (Poch-Massegú et al. 2014) and run off into surface water, contaminating drinking water and aquatic ecosystems (Cameron et al. 2013). Regulatory programs, such as California's Irrigated Lands Regulatory Program, require growers to report their N efficiency practices and account for N inputs and sources on their farms, including soil available N in the root zones (Central Valley Regional Water Quality Control Board 2018). Testing for soil N is an important but often underutilized practice (Central Valley Regional Water Quality Control Board 2018). More widespread and accurate use of soil N testing could reduce input costs and increase yields and quality of crops while reducing N losses.

Abstract

Small-grain crop growers need to match their crops' nitrogen (N) needs with fertilizer applications. This can be challenging because small grains are grown under diverse conditions and their growing season interacts with unpredictable precipitation. Resulting conditions can lead to nitrate-N leaching and runoff losses. More widespread and accurate soil N testing could help growers improve N fertilizer use efficiency, reduce the risk of N loss, and fulfill regulatory requirements. Soil samples from across California small-grain growing regions were tested with a soil nitrate quick test as well as standard laboratory procedures. The quick test is inexpensive and easy to use, and it provides rapid results. A correction factor was developed to convert the quick test values to lab and fertilizer equivalents. The correction factor is based on site-specific soil bulk density and the extracting solution used. An interactive webtool was developed that integrates this information for users. The quick tests provide accurate, real-time estimates of soil nitrate-N in the field to help improve fertilizer use efficiency and reduce N losses.

Author Michael Rodriguez collecting soil samples from a small grain field in the Intermountain Region. UC researchers modified the soil nitrate quick test to help simplify interpretation and increase the accuracy of its results, which could help growers improve fertilizer use efficiency. *Photo:* Taylor Nelsen.

Taking a soil sample at a depth of 1 foot (0–12 inches) with an open auger in moist soil and getting ready to preform a nitrate quick test on the field-moist samples. *Photo:* Taylor Nelsen.

The soil nitrate quick test can be conducted in the field with relatively simple procedures and easily obtained materials, and its nitrate-N results correlate with results obtained by lab analysis (Breschini and Hartz 2002; Hartz et al. 1994; Hartz et al. 2000). This test is a more accessible and inexpensive alternative to laboratory soil N testing and it detects nitrate-N, which is typically the most plant-available form of soil N. The soil nitrate quick test is one of many types of on-site field nitrate tests that have been tested and are available for nitrate-N determination (Tully and Weil 2014). Nitrate moves easily within the soil-water-plant continuum, and concentrations of soil nitrate-N can change quickly in a field along with changes in soil moisture and plant demand. Therefore, the soil nitrate quick test has been especially recommended to inform in-season fertilization management decisions (Lazicki and Geisseler 2016).

Current recommendations for conducting a soil nitrate quick test are based on Hartz et al. (1994) and utilize a 3:1 ratio of a calcium chloride ($CaCl_2$) solution to a homogenized soil sample. Factors such as soil texture (sand, loam, clay) and soil moisture (wet, dry) are used to correct soil nitrate quick test values to soil nitrate-N lab equivalents according to an empirically derived linear equation. Users employ a chart to self-identify their soil properties and convert their soil nitrate quick test value to a soil nitrate-N lab equivalent via a linear constant (Hartz 2010; Smith and Cahn 2019).

Based on more than 300 soil samples taken from 19 site-years across the small-grain growing regions of California, the original method reported by Hartz et al. (1994) has been modified and updated. Changes include: (1) expressing quick test as lab-equivalents via log-linear rather than simple-linear relationship (and then transforming the result back again for ease of interpretation), (2) incorporating statistical uncertainty into the estimation and reporting of quick test values, shown as a margin of error, (3) for the correction factor, using bulk density as a continuous variable rather than using broad categories of soil textures and (4) providing distinct estimates for quick tests conducted with or without calcium chloride in the shaking solution. In addition, a web-tool was developed from these data. This tool provides automated conversion of soil nitrate quick test values to nitrate-N fertilizer equivalent in pounds per acre based on map-enabled, site-specific soil information for California small-grain fields. These updates are intended to simplify interpretation and improve the precision of the soil nitrate quick test, thereby expanding its use in California agriculture.

Soil sampling methods

Samples were taken from the top foot (0 to 12 inches) of the soil profile from 19 different site-years that approximate the range of soil types and associated bulk densities for California small-grains crops (fig. 1). Sampling was conducted according to the principles outlined in Geisseler and Horwath (2016). A total of 327 soil samples were taken and analyzed between 2014 and 2019. Individual locations were often sampled multiple times within a season as soil nitrate levels changed or if there were known fertility gradients. Soil samples were placed in a paper bag and air dried until reaching equilibrium. They were then crushed into small pieces (< 0.2 inches). A separate set of equivalently sampled soils (n = 27) were taken from 10 site-years to compare soil nitrate quick test results using field-moist samples versus soils air dried in the lab.

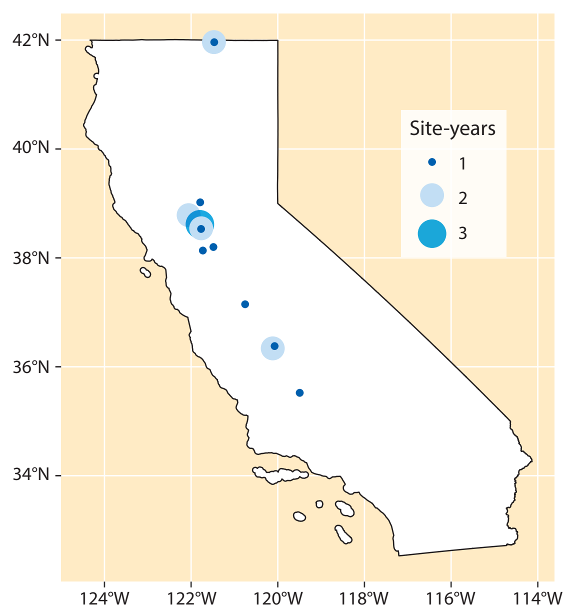


FIG. 1. Locations and the number of site-years at each location where soil samples were taken. Further information about site-years can be found in table 1.

TABLE 1. SSURGO-estimated bulk density, number of samples, number of unique users performing the soil nitrate quick test, the location of the soil sample and the season during which it was sampled

Soil name	Organic matter *(%)*	SSURGO-estimated bulk density *(g/cm^3)*	Number of samples	Number of users	Latitude	Longitude	Season
Tulana silt loam	7.5	0.63	53	4	41.97	−121.47	2014–15
Tulana silt loam	7.5	0.63	46	4	41.97	−121.47	2015–16
Tulebasin mucky silty clay loam	12.39	0.89	51	3	41.96	−121.47	2016–17
Yolo loam	2.26	1.40	36	4	38.52	−121.77	2014–15
Yolo loam	2.26	1.40	29	3	38.53	−121.77	2016–17
Reiff very fine sandy loam	0.75	1.48	6	2	38.53	−121.77	2018–19
Yolo silt loam	2.05	1.42	55	5	38.54	−121.78	2015–16
Yolo silt loam	2.05	1.42	106	5	38.54	−121.78	2017–18
Yolo silt loam	2.05	1.42	1	1	38.78	−121.83	2018–19
Rindge mucky silt loam	17.73	0.77	1	1	38.20	−121.49	2018–19
Panoche clay loam	0.66	1.42	54	4	36.34	−120.12	2015–16
Panoche clay loam	0.66	1.42	2	1	36.34	−120.12	2018–19
Escano clay loam	2.00	1.40	2	1	37.14	−120.75	2018–19
Grandbend loam	1.62	1.45	1	1	39.03	−121.84	2018–19
Rincon silty clay loam	2.00	1.45	60	4	38.78	−122.05	2017–18
Rincon silty clay loam	2.00	1.45	2	1	38.80	−122.05	2018–19
Lerdo complex	1.20	1.46	2	1	35.52	−119.49	2018–19
Diablo-Ayar clays	2.33	1.30	31	3	38.14	−121.74	2016–17
Ciervo	0.90	1.40	1	1	36.39	−120.08	2018–19

Nitrate quick tests

Subsamples were tested for nitrate-N content using commercially available nitrate strips (WaterWorks). A ratio of one part crushed soil was added by volume to three parts distilled water or 0.01M calcium chloride solution for a total of 80 milliliters (ml) of soil solution. Soils were shaken for 3 minutes on a tabletop shaker on high speed. The nitrate-N pad on a strip was tested with a 10 parts per million (ppm) standard solution to ensure that the colorimetric response on the strip matched the 10 ppm nitrate-N box on the color ramp chart. A separate strip from the same bottle was then dipped into the shaken soil slurry, just enough to wet the edge of the nitrate-N pad and allow the solution to wick up the pad. After 60 seconds, the strip was read by the user by matching the color that had developed on the nitrate-N pad to the nitrate-N color ramp chart on the bottle. Bottles of test strips were kept in a cool, dry place until time of use to prevent discoloration and denaturing of the strips. If there was not an exact color match, the user visually estimated the best match for the sample concentration using the values on the color ramp chart. A corresponding subsample was tested for nitrate-N using typical laboratory procedures. Samples were either sent to the UC Analytical Lab or a nitrate-N extraction was performed using 6 grams (g) of crushed dried soil, extracted with 2M postassium chloride following the methods detailed in Doane and Horwath (2003).

The soils used to compare soil nitrate quick test values from field-moist soils and soils air dried in the lab were processed and measured as described above. In addition, in the field, they were tested with a 3:1 ratio of shaking solution to soil, but in larger quantities (300 ml:100 ml). Both calcium chloride and water-only shaking solutions were tested separately. The soil solution was shaken vigorously by hand for 3 minutes.

Another commercially available soil nitrate test was also evaluated (LusterLeaf Rapitest Soil Test Kits) using a subset (n = 34) of the same soils chosen to represent the range of values in the larger dataset. Tests

Researchers reading a soil nitrate quick test. First, the nitrate-N pad was dipped into the shaken soil slurry just enough to allow the solution to wick up the pad. After 60 seconds, the strip was read by the user by matching the color that had developed on the nitrate-N pad to the nitrate-N color ramp chart on the bottle. *Photos:* Taylor Nelsen.

were conducted as described in the product directions, and soil solutions were made with both 0.01M calcium chloride solution and distilled water only.

Soil data

Site-specific bulk density was estimated using the SSURGO database (USDA NRCS 2012; USDA NRCS 2019) based on the geographic coordinates of each soil sample. The variable representing the oven dry weight of the less than 2 millimeters (mm) soil material per unit volume of soil at a water tension of one-third bar was used (approximating field capacity) (USDA 2014). Values used and presented here are a component weighted average of these bulk densities for soil types present in the top foot. It is important to note that bulk density can change due to management and time, and that no direct measurements were made or analyzed in this study. SSURGO-estimated bulk density was used as a covariate in the mixed linear models and as a moderating variable in the conversion of soil nitrate-N lab equivalent to a fertilizer equivalent (i.e., pounds per acre [lb/ac] nitrate-N in the top foot) in the web-tool.

Statistical models

Statistical models with both fixed and random variables (mixed linear models) were used to understand the source of variation resulting from differences in the explanatory variables of soil nitrate-N concentration (measured by the soil nitrate quick test), SSURGO-estimated soil bulk density, extracting solution type, soil moisture status, site-year and user. Repeated measures within a site and/or season (spatial and temporal autocorrelation) were accounted for within the models (nested, random effects). The nonlinear fixed effects (nlme) package in R was used to fit these models (Pinheiro et al. 2020; R Core Team 2020). The significance of each factor and its effect on other factors was tested in relation to soil nitrate-N lab values as the response variable. Both user and site-year were significant factors when tested in a linear model. Because the quick test to lab-value correction equation needs to be valid for any site-year or user, these factors were accounted for as random variables. Soil nitrate-N lab values and soil nitrate quick test values were expressed in logarithmic terms in order to meet model assumptions for analysis and then transformed back to their original values for interpretation. The final model is shown below:

ln(lab measured nitrate-N value + 1) = ln(quick test pad value + 1) * SSURGO-estimated soil bulk density + ln(quick test pad value + 1) * extracting solution + SSURGO-estimated soil bulk density * extracting solution, random = ~ 1| site-year/user

Type-three ANOVA and marginal R^2 values were used to describe the statistical significance of the factors and the percent of variation explained by the model.

Development of a web-tool

An R Shiny tool (Nelsen et al. 2020) was developed that enables users to access the statistical model and SSURGO soil data used in this analysis for their specific location by dropping an interactive map pin at the site of their soil sample. Soil data and the user interface are limited to areas of agricultural production in California (California Department of Water Resources 2014). Users can then enter the soil nitrate quick test value measured, choose whether their test was conducted with a calcium chloride or water-only extracting solution, and adjust the default SSURGO-estimated bulk density, if desired. The web-tool interactively predicts the site-specific soil nitrate-N lab equivalent, fertilizer N equivalent, and an estimate of the margin for error using the results of the mixed linear model reported here. The underlying, open-source code for the web-tool is available at github.com/Grain-Cropping-Systems-Lab/UC-Small-Grain-Soil-Nitrate-Quick-Test-Tool.

Results compared to lab

The soil nitrate quick test was a good proxy for the soil nitrate-N status in small-grain fields. Based on the mixed effects model, quick test values and values measured using standard laboratory methods had a log-linear relationship ($P = 0.02$, $R^2 = 0.83$; fig. 2, table 2).

For water-only shaking solution:

$$\text{lab nitrate-N equivalent} = [e^{1.16 + 1.03\ln(\text{pad value} + 1) - 0.15(\text{bulk density}) - 0.18\ln(\text{pad value} + 1)(\text{bulk density})}] - 1$$

For calcium chloride shaking solution:

$$\text{lab nitrate-N equivalent} = [e^{0.76 + 0.95\ln(\text{pad value} + 1) + 0.19(\text{bulk density}) - 0.18\ln(\text{pad value} + 1)(\text{bulk density})}] - 1$$

The concentration of soil nitrate-N explained 75% of the overall variation in the linear relationship between the quick test and laboratory methods. The SSURGO-estimated bulk density, type of extracting solution, and their interactions explained another 7% of the variation in the linear relationship. Of this 7%, the interaction between the quick test value and the SSURGO-estimated bulk density explained most of the variation. Soils with lower bulk densities required larger correction coefficients ($P < 0.01$) (fig. 2, table 2). This is due to the fact that the soil-to-liquid ratio was established by volume in the quick tests, and soils with lower bulk densities have less mass per unit soil volume. There was also a significant interaction between

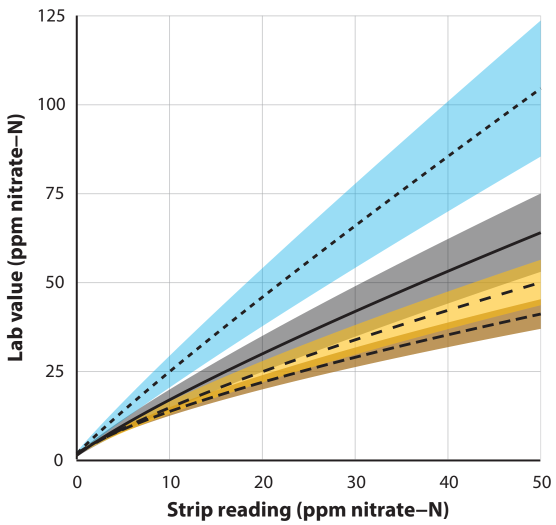


FIG. 2. Linear relationship between the soil nitrate quick test values (ppm) that appear on the WaterWorks color-ramp and the lab equivalent nitrate-N in dry soil (ppm) for the minimum and maximum SSURGO-estimated bulk density tested as well as the standard error that surrounds the estimates. The regression equations are as follows:

For water-only shaking solution:

$$\text{lab nitrate-N equivalent} = [e^{1.16 + 1.03\ln(\text{pad value} + 1) - 0.15(\text{bulk density}) - 0.18\ln(\text{pad value} + 1)(\text{bulk density})}] - 1;$$

for calcium chloride shaking solution:

$$\text{lab nitrate-N equivalent} = [e^{0.76 + 0.95\ln(\text{pad value} + 1) + 0.19(\text{bulk density}) - 0.18\ln(\text{pad value} + 1)(\text{bulk density})}] - 1.$$

TABLE 2. The soil nitrate quick test pad reading (ppm) and the equivalent lab value (ppm) for the different shaking solutions and SSURGO-estimated bulk density at representative values in the range of tested values

Soil nitrate quick test value *(ppm)*	Shaking solution	SSURGO-estimated bulk density *(g/cm³)*	Lab nitrate-N value *(ppm)*	Standard error
0	Calcium chloride	0.7	1.46	0.39
0	Calcium chloride	1.1	1.66	0.23
0	Calcium chloride	1.5	1.87	0.49
0	Water only	0.7	1.87	0.49
0	Water only	1.1	1.70	0.29
0	Water only	1.5	1.54	0.33
5	Calcium chloride	0.7	9.77	1.58
5	Calcium chloride	1.1	9.24	0.79
5	Calcium chloride	1.5	8.73	0.81
5	Water only	0.7	13.43	2.27
5	Water only	1.1	10.95	1.10
5	Water only	1.5	8.89	1.14
10	Calcium chloride	0.7	16.74	2.61
10	Calcium chloride	1.1	15.15	1.26
10	Calcium chloride	1.5	13.70	1.24
10	Water only	0.7	23.93	3.91
10	Water only	1.1	18.76	1.82
10	Water only	1.5	14.66	1.80
20	Calcium chloride	0.7	29.23	4.50
20	Calcium chloride	1.1	25.26	2.13
20	Calcium chloride	1.5	21.82	2.05
20	Water only	0.7	43.67	7.10
20	Water only	1.1	32.81	3.20
20	Water only	1.5	24.59	3.04
50	Calcium chloride	0.7	61.77	9.71
50	Calcium chloride	1.1	50.19	4.50
50	Calcium chloride	1.5	40.75	4.24
50	Water only	0.7	98.46	16.50
50	Water only	1.1	69.64	7.25
50	Water only	1.5	49.17	6.45

Data are estimated from the mixed linear model.

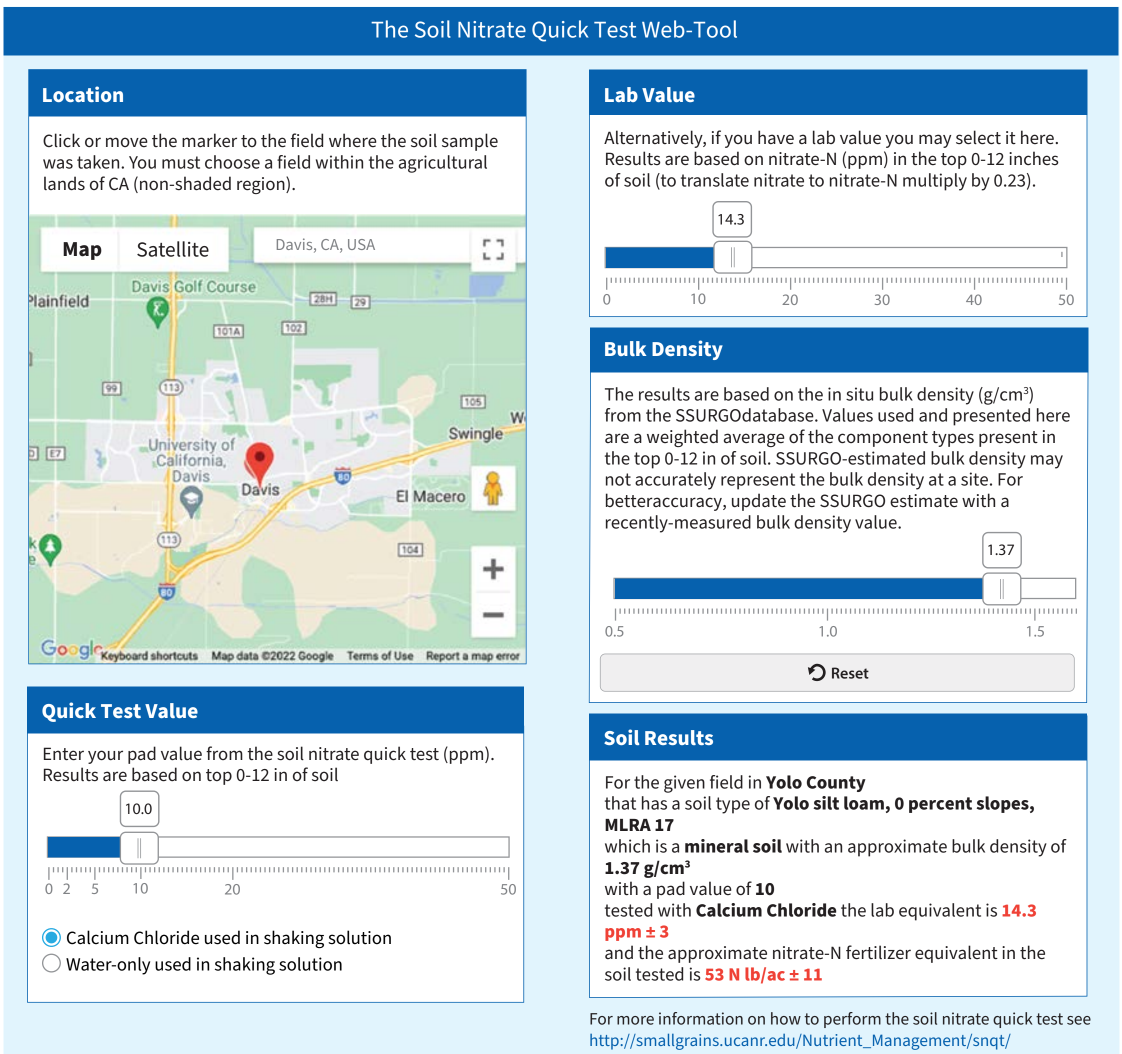
the shaking solution used and the linear correction (P = 0.02) (fig. 2, table 2). Specifically, at higher soil nitrate values, quick tests conducted in a water-only shaking solution required larger correction coefficients than tests conducted in a 0.01M calcium chloride solution. Errors were greater when soils had very low soil nitrate quick test values (< 5 ppm) as well as when lower bulk density soils were tested without calcium chloride in the mixing solution (fig. 2, table 2).

There was no significant difference between nitrate-N measured with quick tests conducted on field-moist soils versus tests conducted on air dried and crushed soils (P = 0.31). As a result, the data presented here do not support translating quick tests to lab equivalent values using a correction factor for moist versus dry soil as in Hartz (2010). The results also indicate that soils can be accurately tested for nitrate in the field without the need to dry and crush the soil. Additionally, there was no interaction between the soil moisture status and the extracting solution used. The effect of SSURGO-estimated bulk density on quick test interactions with soil moisture could not be explored due to the lack of variation in bulk density in this data subset.

The commercially available soil nitrate tests (LusterLeaf Rapitest Soil Test Kit) showed similar linear relationships to standard laboratory values as the quick test (P < 0.01). However, these tests were less accurate (R^2 = 0.59 versus R^2 = 0.83) (data not shown).

Web-tool

Because the relationships between quick test and laboratory equivalent soil nitrate-N values are log-linear and interact with both continuous variables and broad categories (e.g., SSURGO-estimated bulk density and extracting solution), translating a quick test value to fertilizer equivalent is less straightforward than applying a simple linear correction. To address this and simplify the translation and interpretation of quick test values, a web-tool was developed based on the quantitative relationships presented here. The tool allows users to input their site and quick test information, displays the site-specific soil properties, and automatically



An example output of The Soil Nitrate Quick Test Web-Tool in Davis, Calif., with a quick test pad value of 10 ppm.

converts quick test values to laboratory equivalent and fertilizer equivalent values using the conversion of the nitrate-N lab equivalent (ppm) to lb/ac nitrate-N in the top foot. This conversion was accomplished according to the equation below:

$$((\text{nitrate} - \text{N lab equivalent (ppm)})/1{,}000{,}000) \times (\text{bulk density g/cm}^3) \times (30.48 \text{ cm/1 ft}) \times 1 \text{ ft} \times (1 \text{ lb/453.6 g}) \times (929 \text{ cm}^2\text{/1 ft}^2) \times (43{,}560 \text{ ft}^2\text{/1ac}) = \text{nitrate} - \text{N fertilizer equivalent (lb/ac)}$$

The web-tool is available at smallgrain-n-management.plantsciences.ucdavis.edu/snqt/. The underlying open-source code is available at github.com/Grain-Cropping-Systems-Lab/UC-Small-Grain-Soil-Nitrate-Quick-Test-Tool.

Quick test has improved accuracy

The soil nitrate quick test provides an accurate estimate of soil nitrate-N availability. Because soil nitrate-N concentrations can change rapidly with changes in the soil water status of a field, it is important to test soil nitrate-N near to the time when a N fertilizer decision is being made. A soil nitrate quick test value greater than 20 ppm typically indicates there is sufficient soil N for immediate plant needs (Fox et al. 1989) and often indicates that N fertilization is not required. However, soil nitrate information should be paired with other plant N status indicators to provide a holistic picture of crop N sufficiency/deficiency (Bowles et al. 2015).

The purpose of these improvements to the estimation of soil nitrate-N via quick tests is to make soil nitrate testing more accurate and easily accessible to California small-grain growers as well as other farmers, with the goal of reducing N fertilizer costs, improving fertilizer efficiency and minimizing N losses. Because regulations now require growers to report their N efficiency practices, these soil nitrate quick tests can be a very useful tool going forward. CA

T.S. Nelsen (to whom correspondence should be addressed, tsnelsen@ucdavis.edu) is Assistant Specialist, M. Rodriguez is Junior Specialist, E. McCullough is Junior Specialist, S.N. Lewin is Undergraduate Assistant, T. Becker is MS Student, G.G. Rosa is Web Developer, and M.E. Lundy is Assistant Professor of Cooperative Extension in Grain Cropping Systems, Department of Plant Sciences, UC Davis; D. Geisseler is Associate Professor of Cooperative Extension in Nutrient Management, Department of Land, Air, and Water Resources, UC Davis; K. Mathesius is UC Cooperative Extension Farm Advisor for Yolo-Solano-Sacramento counties.

References

Bowles TM, Hollander AD, Steenwerth K, Jackson LE. 2015. Tightly-coupled plant-soil nitrogen cycling: Comparison of organic farms across an agricultural landscape. PLoS ONE 10(6):e0131888. https://doi.org/10.1371/journal.pone.0131888

Breschini SJ, Hartz TK. 2002. Presidedress soil nitrate testing reduces nitrogen fertilizer use and nitrate leaching hazard in lettuce production. HortScience 37(7):1061–4. https://doi.org/10.21273/HORTSCI.37.7.1061

California Department of Water Resources. 2014. Crop mapping 2014. https://data.cnra.ca.gov/dataset/statewide-crop-mapping

Cameron KC, Di HJ, Moir JL. 2013. Nitrogen losses from the soil/plant system: A review. Ann Appl Biol l 162(2):145–73. https://doi.org/10.1111/aab.12014

Central Valley Regional Water Quality Control Board. 2018. Irrigation and nitrogen management plan (INMP) worksheet. Irrigated Lands Regulatory Program - Forms & Templates. www.waterboards.ca.gov/centralvalley/water_issues/irrigated_lands/forms_templates/nitrogen_mgt_plan/inmp_temp_instruct.pdf (accessed Oct. 16, 2020).

Doane TA, Horwath WR. 2003. Spectrophotometric determination of nitrate with a single reagent. Anal Lett 36(12):2713–22. https://doi.org/10.1081/AL-120024647

Fox RH, Roth GW, Iversen KV, Piekielek WP. 1989. Soil and tissue nitrate tests compared for predicting soil nitrogen availability to corn. Agron J 81:971–4. https://doi.org/10.2134/agronj1989.00021962008100060025x

Geisseler D, Horwath WR. 2016. Sampling for soil nitrate determination. http://geisseler.ucdavis.edu/Guidelines/Soil_Sampling_Nitrate.pdf (accessed Oct. 16, 2020).

Hartz TK. 2010. Using the pre-sidedressing soil nitrate 'quick test' to guide N fertilizer management. UC Vegetable Research and Information Center. https://vric.ucdavis.edu/pdf/FERTILIZATION/fertilization&soil_Using%20the%20pre-sidedressing%20soil%20nitrate%20%E2%80%98Quick%20Test%E2%80%99%20to%20guide%20N%20fertilizer%20management.pdf (accessed Oct. 30, 2020).

Hartz TK, Bendixen WE, Wierdsma L. 2000. The value of presidedress soil nitrate testing as a nitrogen management tool in irrigated vegetable production. HortScience 35(4):651–6. https://doi.org/10.21273/HORTSCI.35.4.651

Hartz T, Smith R, Schulbach K, LeStrange M. 1994. On-farm nitrogen tests improve fertilizer efficiency, protect groundwater. Calif Agr 48(4):29–32.

Lazicki P, Geisseler D. 2016. Soil nitrate testing supports nitrogen management in irrigated annual crops. Calif Agr 71(2):90–5. https://doi.org/10.3733/ca.2016a0027

Nelsen T, Rosa G, Lundy M. 2020. The Soil Nitrate Quick Test Web-Tool. Retrieved from http://smallgrain-n-management.plantsciences.ucdavis.edu/snqt/

Pinheiro J, Bates D, DebRoy S, et al. 2020. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-148. https://CRAN.R-project.org/package=nlme

Poch-Massegú R, Jiménez-Martínez, J, Wallis, KJ, et al. 2014. Irrigation return flow and nitrate leaching under different crops and irrigation methods in Western Mediterranean weather conditions. Agr Water Manag 134:1–13. https://doi.org/10.1016/j.agwat.2013.11.017

R Core Team. 2020. R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. www.R-project.org/

Smith R, Cahn MD. 2019. Details on the nitrate quick test. Salinas Valley Agriculture blog. https://ucanr.edu/blogs/blogcore/postdetail.cfm?postnum=4406 (accessed Oct. 16, 2020).

Tully KL, Weil R. 2014. Ion selective electrode offers accurate, inexpensive method for analyzing soil solution nitrate in remote regions. Commun Soil Sci Plan 45:1974–80. https://doi.org/10.1080/00103624.2014.912297

[USDA NRCS] US Department of Agriculture Natural Resources Conservation Service. 2012. SSURGO Data Packaging and Use.

USDA NRCS. 2014. SSURGO Metadata – Table Column Descriptions Report.

USDA NRCS. 2019. Web Soil Survey. https://websoilsurvey.nrcs.usda.gov/ (accessed Jan. 13, 2020).

UNIVERSITY OF CALIFORNIA
Agriculture and Natural Resources

California Agriculture

2801 Second Street
Room 181A
Davis, CA 95618-7779
calag@ucanr.edu
Phone: (530) 750-1223
Fax: (530) 756-1079

Visit us online:

calag.ucanr.edu
twitter @Cal_Ag
www.facebook.com/CaliforniaAgriculture

Online UC ANR webinars and classes

2022 California Naturalist Conference

https://ucanr.edu/sites/2022CalNatCon/

Date: October 7–9, 2022
Time: 8:30 a.m. to 5:00 p.m.
Location: Tahoe City, CA
Contact: anrprogramsupport@ucanr.edu or 530-750-1361

UC Ag Experts Talk: Managing Voles, Rats and Mice in Orchards

https://ucanr.zoom.us/webinar/register/WN_sosKLIlaR_WcyB2FTlvDqA

Date: October 26, 2022
Time: 3:00 p.m. to 4:00 p.m.
Location: Online
Contact: Cheryl Reynolds creynolds@ucanr.edu

8th California Oak Symposium

https://ucanr.edu/sites/oaksymposium/

Date: October 31–November 3, 2022
Time: 8:00 a.m. to 8:00 p.m.
Location: San Luis Obispo, CA
Contact: anrprogramsupport@ucanr.edu or 530-750-1361