California Agriculture

April–June 2021 | Volume 75 Number 2

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

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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.

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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 10,000 print subscribers.

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OUTLOOK

COVER: A robotic harvester manufactured by advanced.farm gently picks a strawberry from a plant on a ranch in Oxnard. *Photo:* Visionary Photography c/o advanced.farm.

Research and review articles

The first SGMA groundwater market is trading: The importance of good design and the risks of getting it wrong

Groundwater markets are a promising tool for basins implementing SGMA, but they are complex, and good design is essential.

Sarah Heard, Director, MarketLab, The Nature Conservancy

Matthew Fienup, Executive Director, Center for Economic Research and Forecasting, and Assistant Professor, California Lutheran University

E. J. Remson, Senior Project Director, California Water Program, The Nature Conservancy

Online: https://doi.org/10.3733/ca.2021a0010

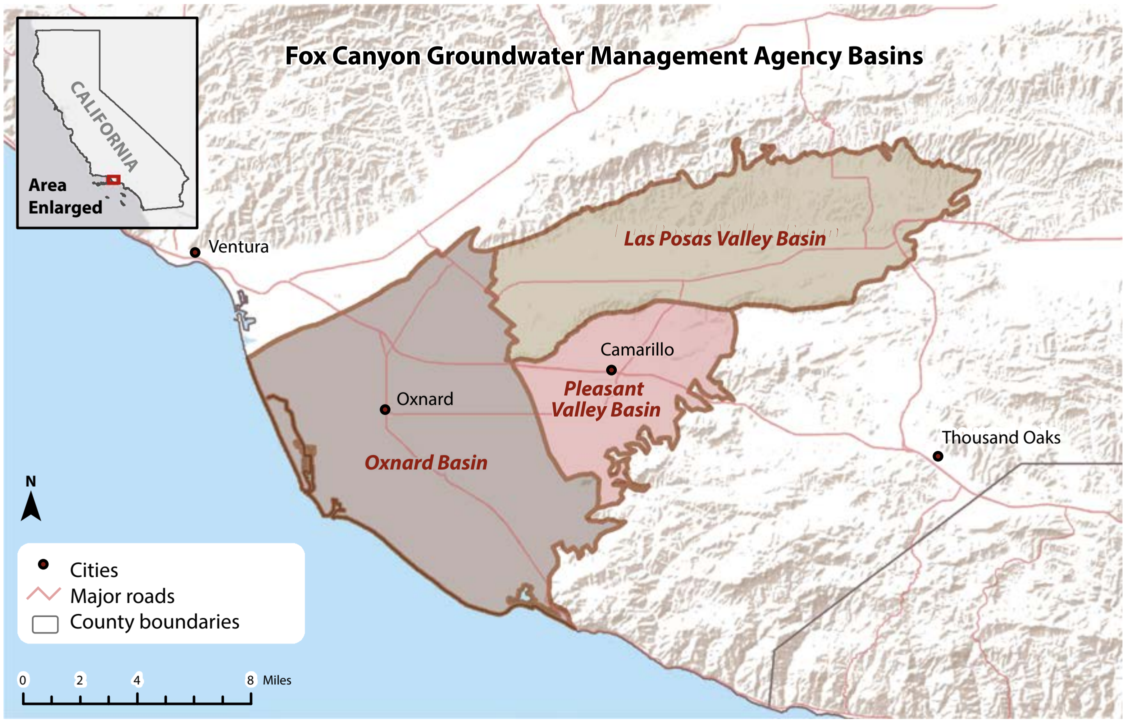
A groundwater market, which caps total pumping within one or more basins, allocates portions of the total to individual users and allows users to buy and sell groundwater under the total cap, is a promising tool for basins implementing California's Sustainable Groundwater Management Act (SGMA). While the benefits of a cap-and-trade system for both groundwater users and regulators are potentially very large, so too are the risks. An electronic bulletin board that introduces buyers and sellers, like craigslist.org, is not a market. Nor is a sophisticated financial application that matches participants and executes financial transactions. A water market is a complex interaction of individuals and institutions — the product of a large number of people, structures, operational mechanisms and rules. Without careful design, a water market can do harm.

Creating a functioning market is not easy. There is no off-the-shelf solution, and there is a lot to get right. The most important — and difficult — elements to get right are the rules and structure, which must be tailored to local conditions. Capping and monitoring pumping, generating buy-in from diverse stakeholders and guarding against cheating and adverse impacts, such as the drying of shallow drinking water wells or of groundwater-dependent ecosystems (GDEs), are also essential. Even with careful design, markets can fall short or cause adverse impacts. And, as the new reality of pumping restrictions sets in, powerful pumpers, largely unregulated before SGMA, will attempt to bend market rules in their favor.

We have lived this experience. Since 2016, we have been in the trenches, developing the Fox

The Fox Canyon groundwater market operates in a large area of Ventura County that includes over 55,000 acres of high-value agricultural land and 500 active agricultural wells. A primary driver of the market is the scarcity of water.

Melinda Kelley



Fox Canyon Groundwater Management Agency (FCGMA) is a Special Act District created by the California legislature in 1982 to address seawater intrusion in three coastal basins in Ventura County. FCGMA was officially designated as the groundwater sustainability agency (GSA) for the three basins with the passage of the Sustainable Groundwater Management Act (SGMA) in 2014. Fox Canyon growers are facing pumping cuts of 40% or more under SGMA.

Canyon groundwater market for three coastal basins, an area known as Fox Canyon, in Ventura County. The first market to be implemented under SGMA, the Fox Canyon groundwater market began trading in early 2020 in the Oxnard basin, which has nearly 200 agricultural wells representing 77,000 acre-feet of pumping. Ventura County is one of the nation's most productive agricultural counties, with $2 billion in agricultural revenue, the majority of which is generated in Fox Canyon (County of Ventura 2019). Water users there are largely groundwater-dependent, and decades of overpumping landed two of the region's basins on the list of 21 SGMA-designated "critically overdrafted basins."

Area growers called for a groundwater market as a tool that would give them flexibility while complying with pumping cuts of 40% or more under SGMA. What began as an open, robust stakeholder process chartered by the Fox Canyon Groundwater Management Agency (FCGMA), led by California Lutheran University (CLU) and supported by The Nature Conservancy (TNC), grew into a multiyear effort to create a model groundwater market under SGMA.

Enabling conditions

Our experience developing the Fox Canyon groundwater market and that of other basins in the United States and overseas indicates that groundwater markets can be a useful tool for achieving basin sustainability, but they are not a good fit for every basin or groundwater sustainability agency (GSA). A number of enabling conditions are necessary to ensure that a groundwater market functions effectively. The Fox Canyon groundwater market benefitted from the four enabling conditions (water scarcity, fixed allocations, agricultural stakeholder support, and capacity and funding) described below.

Water scarcity

Without significant scarcity, a market will not function and is likely not needed. A primary driver of the Fox Canyon groundwater market is the degree of scarcity that agricultural users are experiencing as they implement the SGMA-mandated reduction in pumping. SGMA's requirement that a basin's sustainable yield be determined fixes the maximum amount of groundwater available for the diverse needs of all pumpers in that basin, essentially serving as a cap on total extractions. If the demand for groundwater exceeds the sustainable yield of the basin, reductions in individual pumping are likely required, as they were in Fox Canyon.

Fixed allocations

Clearly defined individual fixed allocations are the first step in the development of a cap-and-trade market; they determine the unit of trade and establish how many units each market participant has to either extract or trade. The sum of the fixed allocations equals the total extraction allowed for the basin in a given year.

FCGMA chose to move from an existing system of pumping allocations that varied by crop type, known as indexed, or efficiency, allocations, to an allocation system based on historical pumping for each well. Water-market participants are assigned pumping allocations in units of 1 acre-foot, to be used or traded during the current water year.

Agricultural stakeholder support

Agricultural stakeholder support for a groundwater market is essential, as farmers represent the largest consumers of groundwater in California. The idea for the Fox Canyon groundwater market originated among a small group of local growers in early 2014, as they were facing the prospect of reduced groundwater supplies as the result of California's drought. Growers recognized that the heterogeneity in both the season and the water demand of the region's crops (ranging from berries, flowers and vegetables to citrus and avocado orchards) created opportunities for a water market (see Fargher 2011). With the help of the Farm Bureau of Ventura County, they brought their ideas to FCGMA's staff and board of directors. Agricultural stakeholders in FCGMA's jurisdiction are well-organized, and the leadership provided by this group was critical.

Capacity and funding

The creation of a water market is a considerable undertaking that requires significant funding and dedicated capacity from GSA staff, participants and partners. During the development of the Fox Canyon groundwater market, TNC secured a Conservation Innovation Grant from the federal Natural Resources Conservation Service, with the support of FCGMA, CLU, the Farm Bureau of Ventura County and local growers. The primary motivation for pursuing the grant was to help implement a sound groundwater sustainability plan (GSP) with an embedded groundwater market that would provide for the needs of both nature and agriculture and that would hopefully serve as a model for others to follow.

The grant provided over $1 million to design and test the market, and to cover the installation of telemetric monitoring hardware to automatically collect pumping data. Without the infusion of funds from the grant, the Fox Canyon groundwater market may not have endured the resource-intensive development and testing phases.

The diversity of the crops in Fox Canyon — flowers, vegetables, berries, citrus, avocado — creates diversity in water demand, which suits a groundwater market.

What good looks like

Establishing a functioning water market involves far more than creating a trading platform. At a minimum, a successful water market requires clear objectives, rules to achieve those objectives and a governance system with resources and the capacity to establish and enforce the rules. For nearly 2 years, a range of stakeholders worked collaboratively to develop the Fox Canyon groundwater market's goals and objectives, rules and operational mechanisms. These were carefully tailored to fit local conditions and the needs of local stakeholders, which is a big part of good design. Well-designed markets in other jurisdictions may look different in some aspects.

Solid groundwater sustainability plan

A GSA wishing to create a water market should create its GSP with the market in mind. A well-designed water market can help achieve the goals of a GSP, but a poorly designed market may undermine the plan. Likewise, superior market design cannot mitigate a GSP's shortcomings. FCGMA created its GSP and water market in parallel. That required significant agency capacity and resources but allowed for iteration between the GSP and the market design so that critical elements of the GSP, such as the sustainable yield and pumping allocations, could support a functioning water market.

Methods to achieve pumping reductions that are overly complex or are not clearly quantifiable on a well-by-well basis may not be compatible with a market. For example, some Fox Canyon growers proposed a rule to allow "borrowing from the future" (pumping beyond a current year's allocation, to be offset by further reductions in future years), but borrowing would undermine the basic structure and function of the market by destroying the price signal upon which individual water-use decisions are made. Without proper attention, plan elements may exclude the possibility of a market. A solid GSP should also establish what is not traded.

Specifically, water within the sustainable yield should provide for human consumption and GDEs; communities and nature should not be required to rely on groundwater markets to meet their water needs. Environmental groups, disadvantaged communities (DACs) and environmental justice organizations throughout California are right to be concerned that water-market activity may be dominated by those with the greatest financial resources or political power, that local groundwater allocations may be allocated disproportionately to these powerful groups and that adverse impacts, such as drying of DACs' shallow drinking water wells or loss of GDEs, may result. These are real risks, and the remedy is a strong GSP that balances economic, environmental and social benefits to ensure compliance with SGMA.

Open, public process

An open process and robust stakeholder input on the creation of a water market are essential in building trust, customizing the structure of the market and market rules and ensuring that stakeholders use the market. FCGMA established a formal stakeholder group, called the Water Market Group, with biweekly meetings that were open to the public, encouraging input on the market design. CLU facilitated the meetings, which typically had 40 to 50 participants, including growers and representatives from water utilities, municipalities with nearly a half-million residents, mutual water companies and environmental organizations.

The meetings focused on learning how water markets function, setting goals for the Fox Canyon groundwater market and establishing trading rules. To help build local knowledge on how water markets work, CLU invited guest speakers with market experience from around the world to address the group. The group also gathered data, case studies and other publications on water markets, and it posted this information on a shared website. A key theme that emerged from the presentations and literature review was the importance of creating a water market that was transparent, fair, easy to understand and low-cost.

After meeting for 7 months, the group unanimously agreed on the outline for the structure and operational mechanisms of a permanent water market as well as on a set of goals and rules to be used in a series of pilot water markets. The group presented these rules to FCGMA's staff and board of directors, and they became the basis for the agency's ordinances that authorized the water market. A series of pilots tested these rules before the market was opened to all agricultural pumpers in the Oxnard basin in February 2020. The group will re-engage, as needed, to address any issues identified and to recommend changes to the rules as the market evolves over time.

Protections against market power

Well-designed markets provide all market participants with equal access to trade and equal opportunity to gain from market activity. This necessitates keeping transaction costs low and creating a fair market that is free of manipulation. Influential parties may attempt to manipulate the price of water and to extract all of the economic gains from trade. They may even seek to exclude others from participating in the market fully. For example, during an early pilot of the Fox Canyon groundwater market, a packer/shipper sought to learn the identities of all growers in the market in order to restrict their participation. Rules and structures

Benefits of well-designed markets

A well-designed groundwater market, in which the price of water is allowed to reflect its true value, has multiple benefits. Notably, a functioning market is a cost-effective tool for achieving SGMA's mandate of sustainable management, driving the reallocation of pumping within a basin to the highest-value uses. The ability to trade motivates users to conserve scarce groundwater, invest in water use efficiency and develop new water supplies, like recharged wastewater — all of which contribute to basin sustainability. The largest benefits typically occur in regions with both water scarcity and variable water demand, seasonal fluctuations in water availability, a large number of interconnected water users with varying demands and degrees of flexibility, agricultural water users who are exposed to the risks that accompany national and global markets, and increasing demands for urban and environmental water (Fargher 2011).

Markets benefit their users by allowing greater flexibility than command-and-control schemes do. For example, growers can generate revenue when fallowing fields and avoid penalties for pumping beyond their allocation by purchasing additional water on the market. In the Fox Canyon groundwater market's first year of trading, a grower avoided nearly $350,000 in surcharges by purchasing the water for less than 15% of that figure. Municipal and industrial users can turn to markets to purchase additional supplies and to sell surplus supplies, like recharged wastewater, to recoup capital costs. Water trading has proven successful in supporting agricultural productivity in a number of settings, from Australia and South America to the Western United States (Fargher 2011; Hearne and Easter 1997).

A well-designed market can also benefit sensitive resources. Special rules can avoid undesirable impacts in areas that need protection against overpumping. In Fox Canyon, pumping was reduced in one of the most vulnerable areas — a pumping trough — without top-down regulatory restrictions that differentially impacted pumpers in a sensitive area. Other sensitive areas can also be spatially delineated, such as groundwater-dependent ecosystems (GDEs) and shallow drinking water wells for rural and disadvantaged communities (DACs). Water markets have enhanced municipal water security and maximized environmental benefits in areas as diverse as Australia and the Western United States (Fargher 2011; Garrick et al. 2009; Garrick et al. 2011).


E. J. Remson

Biweekly meetings facilitated by CLU brought together growers and representatives from water utilities, municipalities, mutual water companies, and environmental organizations. A solid agreement emerged that the market needed to be transparent, fair, easy to understand, and low cost.

Sarah Heard


Sarah Heard

Accurate water monitoring is a first-order concern to ensure functioning groundwater markets. Meters, in place on Fox Canyon groundwater wells since the 1980s, track water use, *left*. To prevent cheating and ensure accurate data collection, Fox Canyon growers opted for universal telemetric sensors that attach to meters and stream pumping data real-time, funded by a grant from the Natural Resources Conservation Service, *right*.

designed by the Water Market Group, including a single, central exchange and anonymized trading, succeeded in preventing unfair influence.

Equal access to the Fox Canyon groundwater market was addressed by implementing a formal, centralized market structure that uses a private exchange administrator, that is, a private organization that is independent of the GSA. Stakeholders felt strongly that the exchange administrator should be independent of the GSA and also that it should be local and nonprofit and not have a financial or other stake in the reallocation of groundwater pumping.

The goal of a fair market was addressed by establishing an anonymized market and blind, algorithmic matching. Bids and offers are submitted and matched, and transfers of pumping are executed and reported to the GSA, all without market participants ever knowing who the counterparties are. And yet the process is transparent and accountable.

Mitigation of adverse third-party impacts

Well-designed markets must anticipate and mitigate the risks of adverse third-party impacts. Market transfers can inadvertently create areas of concentrated pumping in the basin that can result in lowered water levels and a decline in water quality, which, in turn, may adversely impact surface water flows, GDEs and other local pumpers. The drinking water supplies of DACs may be particularly vulnerable.

Mitigation starts with a basin's allocation system, for example, ensuring adequate water for GDEs and DACs. Once allocations have been established, foresight is required to anticipate when a particular transfer of allocation might adversely impact third parties. Specific market rules are required to prevent these unintended impacts. It may also be necessary for market rules to adapt over time to address unintended impacts.

Special management areas (SMAs) are delineated geographic areas established by the GSP to address the risk that trading may negatively impact groundwater quality or levels. Rules can be implemented to restrict the volume or direction of water transfers within an SMA. SMAs have broad applications and can be used to address adverse impacts to surface water flows, GDEs and DACs.

The Fox Canyon groundwater market includes two SMAs; one is an area of seawater intrusion, the other is a local pumping trough. The Fox Canyon groundwater market's rules stipulate that pumpers in an SMA may purchase additional water only from another pumper within the SMA but that they may sell either to a pumper within the same SMA or to a pumper outside of both SMAs. The goal of these directional restrictions is to ensure that transfers of pumping allocation do not result in a net increase in pumping within an SMA. In practice, the use of this tool has resulted in a market transfer of pumping out of one SMA into a healthier part of the basin.

Directional trading is one of a number of approaches used to protect SMAs. Exchange rates, or trading ratios, whereby one unit of pumping outside an SMA is traded for less than a unit of pumping within an SMA, have also been used in other markets.

Accurate, reliable monitoring of extraction

Accurate water-use data is critical to achieving sustainable groundwater management. Errors in the measurement of water use have been shown to produce large economic losses for farmers and to undermine policies to limit adverse impacts on the environment and other water users. Choosing monitoring technology involves trade-offs, notably between implementation cost and accuracy; one of the least costly options, satellite remote sensing, has been shown to produce large measurement errors (Foster et al. 2020).

A water market also needs accurate water-use data to ensure that participants trade only unused water allocations and that no exceedances of pumping allocations result from trading. Accurate monitoring is a first-order concern for water market participants. Any underreporting of water use, or other form of cheating, devalues allocations available for trade on the market and undermines progress toward achieving a basin's sustainable yield, potentially resulting in further cuts down the road.

In Fox Canyon, growers were deeply concerned that pumping be accurately measured and so they proposed universal telemetric monitoring of groundwater extraction with automated reporting. Fox Canyon pumpers now use cellular-based telemetry attached to individual meters; this system broadcasts pumping data to a cloud-based data portal. The portal automatically submits monthly pumping totals to the GSA. Hardware approved for use in Fox Canyon also includes validation measures designed to prevent cheating.

In the early 2000s, FCGMA's staff and board of directors had discussed a requirement that all agricultural wells employ electronic monitoring and automated reporting, using early telemetry hardware, but protest from the agricultural community was so strong that the plan was abandoned. The new Fox Canyon groundwater market altered incentives, making universal telemetric monitoring of extractions not just politically feasible but imperative to agricultural water users.

Market testing: learning and adaptation

Water markets are complex. They involve an almost dizzying interaction of individuals, institutions, actions and reactions. Critical questions arise for those implementing markets: How will progress be evaluated? Does the market work as intended? What are the unintended consequences? Does the market structure adapt to new information as it becomes available? Answering these questions requires a humble approach: starting simple, testing early and often and creating a market structure that allows for adaptation over time.

During the design phase, the Water Market Group recommended testing the market, with a definitive starting and ending point, to ensure that the market functioned as intended. The goal was to test the rules and any intended market outcomes while also allowing FCGMA and market participants to discover and address any unintended consequences of trading. A series of sequential pilots was implemented. The phases included a demonstration project for the telemetric monitoring and automated reporting system, stress testing of the market rules and the electronic trading platform and trading between pumpers in the largest basin. Numerous issues were identified and addressed prior to full market implementation. Had these issues not been addressed early, they may have forever undermined participants' faith in the market and its ability to function.

Why markets fail

The benefits of water trading can be reduced by a number of factors, including regulatory uncertainty, such as changes to rules or allocations that undermine participants' ability to trade (Grainger and Costello 2014); high transaction costs (Crase et al. 2000; Donoso 2006); the use of market power by one or more participants to restrict access to the market or to manipulate the price of water (Ansink and Houba 2012; Brozović 2016; Bruno and Sexton 2020) and adverse impacts to third parties (Heaney et al. 2006).

As the most important and most common sources of friction in markets, transaction costs and market power warrant special consideration. Transaction costs, which in extreme situations can be greater than the cost of the water itself, include the costs of bringing together willing buyers and sellers of water, negotiating the price and other terms of a trade, validating ownership of the water use right, legalizing the contract, enforcing contract provisions and gaining regulatory approval for a transfer (Crase et al. 2000; Donoso 2006).

A participant exerting market power might benefit from driving the price artificially low (if they plan to buy) or artificially high (if they plan to sell). Even a small degree of exerted market power can cause sizeable impacts (Bruno and Sexton 2020). It can deter potential users, either directly or indirectly, from participating and can increase the risk of the market languishing or even collapsing.

Trust in and the perceived fairness of a market are particularly important. In a number of markets involving agricultural water users, farmers have been shown to forego participation, despite direct financial benefits, due to a lack of trust. Historical mistrust of regulators and other actors, along with fear that the benefits and responsibilities are not equally distributed, are primary causes of an unwillingness to participate (Breetz et al. 2005).

A call to action

Groundwater markets existed in California, and elsewhere, long before SGMA. But with SGMA's new mandate to achieve basin sustainability across large parts of the state, interest in groundwater markets is growing. According to California's Department of Water Resources, 20 of 46 GSPs submitted to date include a groundwater market as a strategy or management action. Markets are complex by their very nature and have steep learning curves. We have learned firsthand the importance of careful design, how much there is to get right and how much work is involved.

If markets are to be a successful tool in complying with SGMA, GSAs will need support and accountability. If not, too much will be left hanging in the balance. Specifically, we recommend:

1. **A standardized framework.** Without the support of a guiding, standardized framework on "what good looks like," the risks of market failure and adverse impacts are too high. Currently, GSAs may develop markets as they wish; there are no required elements, like stakeholder involvement or accurate measurement. The standardized framework should articulate the essential elements of a well-functioning market under SGMA — in a broadly applicable, rather than prescriptive, way — so that any GSA could use it to tailor the design of a market to its

If a groundwater market is trusted by its participants and well used, it's a promising tool to rebalance an overdrawn aquifer. But markets need support and accountability to make sure they succeed.

basin's conditions. One possible framework might be an accreditation program administered by an independent body composed of experts in market design.

2. **Resources for market development.** In addition to a standardized framework, funding and technical expertise are essential to produce functioning markets. State funding for additional staff capacity and outside expertise would help guard against the development of poorly designed markets that are set up to fail.
3. **Accountability.** Once they have the right ingredients to produce "good markets" — a standardized framework, funding and technical expertise — GSAs must then be held accountable for the outcomes of implementing groundwater markets. Markets don't end at design; they need regular evaluation to ensure that they function as intended. They should be adapted if they underperform or cause adverse impacts. Evaluating and reporting on the outcomes of implementing groundwater markets are not currently required but should be required going forward.

Groundwater markets hold great promise as we seek to rebalance our aquifers, but only if they are recognized for the complex tools that they are. It is still early enough to ensure that groundwater markets take shape in a way that helps implement, rather than undermines, SGMA's objectives. The risk of failure is great, and, if we fail, we may see little progress on SGMA or, worse, an exacerbation of already dire aquifer conditions. CA

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RESEARCH ARTICLE

Robotic strawberry harvest is promising but will need improved technology and higher wages to be economically viable

An analysis of harvest efficiencies and wage rates suggests that adoption of robotic harvesters is not yet economically feasible for large strawberry growers.

by Timothy Delbridge

Online: https://doi.org/10.3733/ca.2021a0009

Recent developments in California state agricultural labor policy, along with the aging agricultural workforce and declines in new-immigrant arrivals, have increased the urgency with which agricultural producers are seeking new labor-saving technologies (Martin 2017, 2018). Even prior to state legislation that increased the minimum wage and reduced the thresholds for overtime pay, real farm wages were rising and the number of farmworkers in the state were falling. From 2003 to 2017, the number of farmworkers employed in California fell by 32% (Bampasidou and Salassi 2019). The potential impact of automated harvest technology designed for strawberry production is particularly high given the labor intensity of the crop, the demanding nature of the work and the high cost associated with manual strawberry harvest.

The strawberry industry has responded to these developments with several separate but related efforts designed to reduce the industry's labor needs. Adoption of harvest aide equipment, which reduces the amount

Abstract

While the prospect of robotic harvest in strawberry production has received much attention within the strawberry industry and the popular press, there is little available information on the economic feasibility of this technology. It is not clear how close the industry is to being able to profitably adopt robotic harvest systems; also unclear is the relative importance of wage rates, robotic harvest efficiencies and machinery field speeds on the adoption threshold. This study aims to clarify these issues by estimating the net income to strawberry production under robotic harvest scenarios, and comparing the values to standard enterprise budgets for strawberry production in California under different wage rates for harvest labor. Results confirm that robotic harvest remains economically unviable under current wage rates and the field speeds and harvest efficiencies achieved by leading robotic harvest development teams. However, results indicate that with expected increases in wage rates in the coming years, and with modest improvements in the technical parameters, use of robotic systems will likely become profitable in some form.

Robotic picking arms on a Harvest Croo Robotics automatic harvester. The speed and accuracy with which robotic harvesters can pick ripe strawberries is critical to their economic feasibility.

G.J. Holmes, Cal Poly Strawberry Center

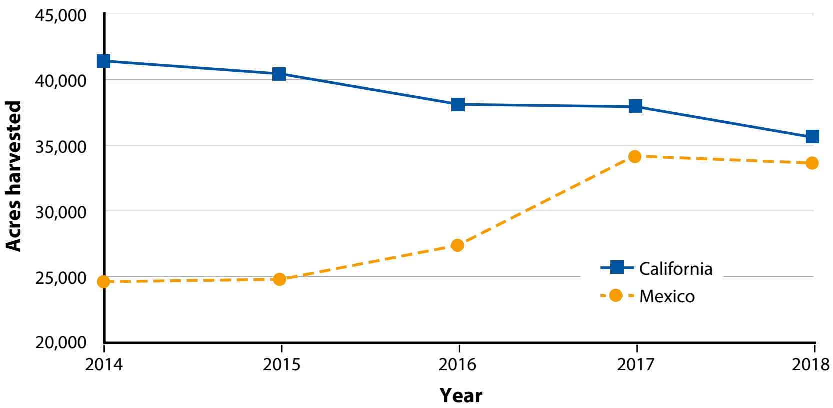

FIG. 1. Harvested strawberry acreage in California and Mexico from 2014 to 2018. *Sources:* USDA-NASS and SIAP databases.

of time that workers spend bringing fruit to the field edge, has increased the speed of manual harvest. There have also been recent pilot programs to develop table-top production systems, which could expand the labor pool from which growers draw, and potentially increase the speed at which workers can harvest fruit (Karst 2018). Though not a technological fix, some production has shifted to Mexico, where labor challenges are less severe. As shown in figure 1, there has been increased planted acreage in Mexico as acreage has contracted in California (SIAP 2019; USDA-NASS 2019).

These initiatives may have decreased the strawberry industry's labor needs on a per unit basis, but the prospect of fully robotic harvest technology remains the potentially transformative development that attracts much attention inside and outside the industry. Individual strawberry growers and shippers have invested heavily in robotics companies in recent years and many widely distributed popular press outlets have published articles on the development and potential impacts of robotic strawberry harvest (e.g., Mohan 2017; Paquette 2019; Seabrook 2019).

Despite this attention and excitement, there has been no publicly available economic analysis of the robotic harvesters currently being developed for in-field strawberry production. It is not clear how profitable robotic systems would be in their current form or how different wages or technical parameters impact the relative profitability of manual and robotic harvest systems. This article aims to fill that gap by developing economic models of strawberry production in California assuming harvest efficiencies and operational strategies of the industry's leading robotic developers.

Any economic analysis of novel and developing technologies faces difficulty in forecasting operating costs and field efficiencies with perfect accuracy. The case of robotic harvest in strawberry production is particularly challenging, as there is no data on the use of these robotic harvest systems in commercial strawberry operations. This study makes assumptions on harvest efficiencies and operating costs, informed by individual interviews with robotic industry leaders, but also presents ranging analysis so that the reader can better understand for themselves where the industry may lie relative to the "robotic-harvest tipping point" at which adoption of these systems will become widespread.

Methods and assumptions

This analysis uses the enterprise budgets developed for conventional strawberry production in Monterey and Santa Cruz counties (Bolda et al. 2016) as a baseline manual-harvest scenario, and then builds on this framework with a range of piece rate harvest costs, robotic harvest efficiencies and robotic harvest speeds. This method is designed to show the expected profitability of the current state of the art in automated harvest, and the degree to which different values of these parameters impact economic feasibility of robotic harvest systems. The primary metric by which economic feasibility is measured is pre-tax net income per acre, after all harvest costs and robotic harvest machinery costs are considered. Expenses that are unrelated to harvest, or that would not be impacted by the adoption of a robotic harvest system, are included in net income calculations, but assumed unchanged from those outlined by Bolda et al. (2016). While the details of this enterprise budget will certainly differ from the experience of individual strawberry growers on the Central Coast of California, this analysis illustrates the impact of the potential adoption of robotic harvest, relative to the baseline manual harvest scenario.

Strawberry production systems and supply chains are complex, and there are several simplifying assumptions that must be made. In particular, this analysis assumes that there are no changes to strawberry varieties, row spacing or other production considerations aside from the harvest method. It is certainly possible that further research will identify cultivars that are better suited for robotic harvest than those popular today, and this would improve the profitability outcomes of the robotic harvest scenario relative to those presented here. While robotic harvest technologies are being developed with the goal of being implementable in existing fields, with traditional row spacing and bed sizes, it is possible that planting practices will be altered in the future to better accommodate advances in robotic harvest systems. It is difficult to anticipate the impacts of such changes, and this study makes no attempt to incorporate these possibilities. Finally, there is often speculation that robotic harvest solutions may impact fruit quality, or require innovations in packaging and shipping practices. Again, this study does not consider these possibilities and estimates the profitability of robotic harvest assuming no impacts to quality or supply chains of fresh market strawberries.

As discussed above, there are no public data on the actual costs of purchasing or operating robotic harvest systems, or on the harvest efficiency and field speed parameters that are crucial for estimating economic feasibility. Moreover, although there are several teams working to bring robotic harvest to California strawberry producers, it is not clear what the market for automated harvest solutions will look like in the coming years. Given the lack of detailed information and in-field performance data, the challenge is to present an analysis that is specific and precise enough to be informative, but general and flexible enough to cover a large segment of the uncertain outcomes in the robotic harvest field. The parameters assumed for this analysis, and the alternative harvest methods and business models used in the robotic harvest scenarios, are based on individual interviews with executives at Harvest Croo Robotics (Plant City, Fla.) and Agrobot (Oxnard, Calif.) in early 2019. The analyses are not meant to reflect these technologies or companies specifically, but rather a range of outcomes that is most likely to occur when growers begin to adopt robotic harvest technology.

Single robotic pass vs. manual cleanup

Two robotic harvest use scenarios are considered: the first assumes a single pass with the robotic harvester, while the second assumes that a human crew will follow to pick fruit that the harvesting machine missed. The lower the harvest efficiency of the robotic harvester, the more fruit that would be left for the supplementary manual pick. There is some disagreement in the industry whether a secondary manual pick would be cost-effective, given that the piece rates offered to incentivize workers to harvest behind the robot might be exceedingly high. In the current system, piece rates tend to increase when fruit loads are low, and workers cannot fill a tray as quickly as during peak production periods (Hill 2019; Wu et al. 2017). Since it is not clear exactly how much would need to be paid on a per-tray basis to workers following a robotic harvester, this analysis includes results reflecting piece rates of 150% and 200% of the piece rate for traditional manual harvest.

A key assumption must be made with respect to the harvest speed attained by a robotic harvester. The overall harvest speed depends on the number of robotic arms on the unit as well as the average harvest speed for each robotic arm. Additional arms added to the unit allow a higher overall field speed (i.e., acres per hour), but also increase the probability that an individual arm has to wait, unused, for the other arms to finish picking the proximal fruit. While the technical details are different for each development team, this analysis assumes 16 harvest arms and a range of harvest speeds, per arm, from one berry every 7 seconds to one berry every 19 seconds. Assuming 15 berries per 1-pound clamshell and eight clamshells per tray, this represents a range of 25 to 69 trays per hour.

Another key assumption relates to the percentage of berries that are successfully harvested by the robotic system, referred to here as the harvest efficiency. Several factors impact robotic harvest efficiency, including the thickness of vegetative growth of the plant, the design of the harvesting arm and the accuracy of the different vision system and fruit identification software technologies used by the firm in question. Harvest efficiencies will vary across the growing season as field conditions evolve and may by impacted by strawberry cultivar and other factors. This study considers harvest efficiencies from 40% to 90% of the human harvest volumes. That is, at the highest 90% level, the robotic harvest would successfully harvest 90% of the 7,000 trays per acre total production assumed in the 2016 cost and return study that is serving as our benchmark.

Assumptions of equipment cost

When considering the total cost of agricultural equipment, one must consider the machinery purchase price, interest rates, repair costs and operating costs. Total annual machinery costs are calculated by applying conservative purchase price and repair estimates to a commonly used machinery cost calculator (Edwards 2019). This analysis uses a purchase price of $500,000, with a 10-year useful life, operator and fruit packing cost of $50 per hour and an annual repair cost of $50,000. The interest rate is assumed to be 4.25%, which is consistent with Bolda et al. (2016). Under these assumptions, the cost of labor to operate the robotic harvester and pack harvested fruit would be between $0.73 and $1.98, depending on field speed (7 to 19 seconds per berry per arm). With other operating and ownership costs included, full robotic harvest costs range between $1.70 and $5.20 per tray, depending on the speed and efficiency (successful pick of 40% to 90% of ripe berries) of the harvester.

As with any costly equipment purchase, the annual hours of use has a significant impact on the per-unit cost of ownership. As such, it will be critical for those operating these machines to fully utilize the equipment, so that the ownership costs are spread out over as

Exterior view of an Advanced Farm Technologies robotic strawberry harvester. Advanced Farm is one of a handful of companies currently working with strawberry growers to test and refine robotic harvest technologies.

TABLE 1. Cost of production and net income for traditional manual harvest system

Piece rate*	Cost of production (per tray)	Net income (per acre)
$1.70	$9.15	$5,981
$1.80	$9.29	$4,987
$1.90	$9.43	$3,992
$2.00	$9.57	$2,998
$2.10	$9.71	$2,003
$2.20	$9.86	$1,009
$2.30	$10.00	$14
$2.40	$10.14	−$980
$2.50	$10.28	−$1,974
$2.60	$10.42	−$2,969
$2.70	$10.57	−$3,963

* Piece rate is the per-tray wage paid to workers and does not include indirect employment costs.

TABLE 2. Net income per acre under different piece rate and robotic harvest efficiency values for robotic harvest scenario number 1 (no secondary hand harvest)*†‡

Piece rate§	Robotic harvest efficiency					
	40%	50%	60%	70%	80%	90%
$1.70	−$12,477	−$9,127	−$5,819	−$2,875	$70	$3,014
$1.80	−$12,540	−$9,259	−$6,022	−$3,170	−$318	$2,533
$1.90	−$12,604	−$9,391	−$6,225	−$3,466	−$706	$2,053
$2.00	−$12,667	−$9,523	−$6,428	−$3,761	−$1,094	$1,572
$2.10	−$12,731	−$9,655	−$6,631	−$4,057	−$1,482	$1,092
$2.20	−$12,794	−$9,787	−$6,834	−$4,352	−$1,870	$612
$2.30	−$12,857	−$9,919	−$7,037	−$4,648	−$2,258	$131
$2.40	−$12,921	−$10,051	−$7,240	−$4,943	−$2,646	−$349
$2.50	−$12,984	−$10,183	−$7,443	−$5,239	−$3,034	−$830
$2.60	−$13,048	−$10,315	−$7,647	−$5,534	−$3,422	−$1,310
$2.70	−$13,111	−$10,447	−$7,850	−$5,830	−$3,810	−$1,791

* Shading indicates that the per-acre net return to robotic harvest is greater than that to a typical manual harvest system.
† Assumes a 10-second per berry pick time.
‡ Machinery cost calculations based on Edwards (2019) and assume $500,000 purchase price, 10-year useful life, 4.25% interest rate, $50 per hour operator labor cost.
§ Piece rate is the per-tray wage paid to workers and does not include indirect employment costs.

A manual harvest crew picking strawberries for fresh market sale. While it is difficult to match the speed and accuracy of human harvesters, rising labor costs make robotic alternatives increasingly attractive.

many trays as possible. Interviews with robotic harvest developers indicate that, given the seasonal fluctuation in fruit load on strawberry plants, more acres will be covered by robotic harvesters in the early and late parts of the season than in the peak production months. In order for the robotic harvester to be fully utilized, the scenarios assume a 50-acre field, with human harvest crews brought on in peak months when the robot will not be able to cover all 50 acres. With this model, the percentage of total fruit volume that is harvested robotically on this 50-acre field fluctuates from 10% in the most pessimistic speed and efficiency assumptions to 50% with the most optimistic assumptions. While the ownership costs per unit harvested could potentially be reduced with transport of the robotic harvester from region to region, or in some regions by scheduling both fall and summer planting to smooth out seasonal fluctuations, this study assumes the April to October harvest season and monthly production volumes presented by Bolda et al. (2016) for the Salinas-Watsonville region (an Excel-based version of all calculations is available from the author on request).

Manual harvest more profitable than robotic

The two robotic harvest scenarios are compared to the baseline case of a typical manual harvest scenario as represented by Bolda et al. (2016). While the cost of production, and the cost of harvest labor in particular, motivates the development of robotic harvest systems, the focus of this analysis is on the pre-tax net income per acre of strawberries grown. This allows for the consideration of reductions in revenue in the first robotic harvest scenario, which is driven by the lower harvest efficiencies expected with the robotic systems.

Table 1 shows the net income per acre of a manual harvest system at different wage rates. Table 2 shows the net income per acre for robotic harvest at manual harvest piece rates ranging from $1.70 to $2.70 per tray, and robotic harvest efficiencies from 40% to 90% of the human harvest volumes. All other parameters and assumptions are constant across these scenarios and discussed in the previous section. The outcomes in table 2 reflect a single pass of the robotic harvester, with no manual follow-up pick. However, net income decreases as harvest piece rates rise because human workers are employed in all robotic harvest scenarios during times of peak production. These workers do not follow the harvesting robot, but rather operate as typical human harvest crew on acreage that the robotic harvester cannot get to when fruit loads are high.

The shaded values in table 2 are those piece rate and efficiency combinations for which the robotic harvest results in a higher net income than the status quo manual harvest system. The robotic harvest system, without human workers carrying out a second harvest pass, only outperforms manual harvesters at robotic harvest efficiency rates at 80% or above, and at manual harvest piece rates higher than those currently faced by growers. It is also worth noting that, while higher wage rates make robotic harvest more attractive, they also reduce the overall profitability of strawberry production if prices remain constant as assumed in this study. The degree to which strawberry prices will evolve over time is unknown, and depends on California planted acreage and yields, and production levels in other strawberry growing regions.

Table 3 shows the per-acre net income assuming a $1.90 per tray piece rate, but with varying robotic harvest field speeds and harvest efficiencies. The speed of harvest per berry for each robotic arm is varied from 7 to 19 seconds and harvest efficiency is again varied from 40% to 90% of human harvest crews. The only combination of these parameters that results in robotic harvest outperforming manual crews is the highest speed and highest efficiency considered. If the assumed wage for manual workers was increased, robotic harvest would be relatively more attractive, but this result shows that significant technological advances are likely necessary before robotic harvest can outperform manual workers on the basis of net return per acre.

Table 4 shows the per-acre net income assuming a 70% harvest efficiency, which is in line with optimistic estimates of the state of leading robotic harvest technology in 2019, and varies harvest piece rate and the robotic harvest speed. Again, only the fastest harvest speeds and highest wage rates result in net incomes that are higher than those estimated for manual harvest systems at the same wage rates. This shows that unless the robotic harvest efficiency can surpass 70% of the human workers' performance, wages will need to increase by 50% before robotic systems will be viable.

A second robotic harvest scenario is considered in which the robotic pick is followed by a supplementary manual pick to collect fruit that the robotic harvest arm missed. Table 5 shows the net income per acre under this scenario when piece rate and harvest efficiency are varied. The table 5 results reflect an assumption that the workers in the follow-up harvest receive a piece rate equal to 150% of the market rate. This table should be compared to table 2, which shows the same parameter assumptions without the follow-up harvest. The supplementary harvest is attractive at lower wage rates and the full range of harvest efficiencies. Perhaps surprisingly, the follow-up harvest results in higher net income than the scenario with only the single robotic pass in most wage rate and efficiency combinations.

Since it is not known what wage rate would have to be offered to workers for a secondary manual harvest, table 6 presents net incomes per acre for this scenario with an assumed piece rate for supplementary harvest that is double the market piece rate for a traditional

TABLE 3. Net income per acre under different robotic harvest speed and harvest efficiency values for robotic harvest scenario number 1 (no secondary hand harvest)*†‡

Harvest speed *(seconds/ berry)*	Robotic harvest efficiency					
	40%	50%	60%	70%	80%	90%
7	–$10,966	–$7,392	–$3,970	–$757	$2,455	$5,407
9	–$12,223	–$8,885	–$5,672	–$2,662	$97	$2,857
11	–$13,031	–$9,818	–$6,895	–$4,136	–$1,377	$1,382
13	–$13,715	–$10,713	–$7,954	–$5,195	–$2,435	$324
15	–$14,283	–$11,524	–$8,765	–$6,006	–$3,246	–$487
17	–$14,930	–$12,172	–$9,413	–$6,654	–$3,894	–$1,199
19	–$15,469	–$12,711	–$9,952	–$7,193	–$4,486	–$1,820

* Shading indicates that the per-acre net return to robotic harvest is greater than that to a typical manual harvest system.
† Assumes a $1.90 per-tray piece rate for manual harvest.
‡ Machinery cost calculations based on Edwards (2019) and assume $500,000 purchase price, 10-year useful life, 4.25% interest rate, $50 per hour operator labor cost.

TABLE 4. Net income per acre under different piece rate and robotic harvest speed values for robotic harvest scenario number 1 (no secondary hand harvest)*†‡

Piece rate§	Harvest speed *(seconds/berry/arm)*						
	7	9	11	13	15	17	19
$1.70	–$400	–$2,149	–$3,481	–$4,441	–$5,180	–$5,773	–$6,269
$1.80	–$579	–$2,406	–$3,809	–$4,818	–$5,593	–$6,213	–$6,731
$1.90	–$757	–$2,662	–$4,136	–$5,195	–$6,006	–$6,654	–$7,193
$2.00	–$936	–$2,919	–$4,464	–$5,571	–$6,418	–$7,094	–$7,655
$2.10	–$1,115	–$3,175	–$4,791	–$5,948	–$6,831	–$7,534	–$8,117
$2.20	–$1,294	–$3,432	–$5,119	–$6,325	–$7,244	–$7,975	–$8,579
$2.30	–$1,473	–$3,688	–$5,446	–$6,701	–$7,657	–$8,415	–$9,041
$2.40	–$1,651	–$3,944	–$5,774	–$7,078	–$8,069	–$8,855	–$9,503
$2.50	–$1,830	–$4,201	–$6,101	–$7,455	–$8,482	–$9,296	–$9,965
$2.60	–$2,009	–$4,457	–$6,429	–$7,831	–$8,895	–$9,736	–$10,428
$2.70	–$2,188	–$4,714	–$6,756	–$8,208	–$9,308	–$10,176	–$10,890

* Shading indicates that the per-acre net return to robotic harvest is greater than that to a typical manual harvest system.
† Assumes a 70% robotic harvest efficiency.
‡ Machinery cost calculations based on Edwards (2019) and assume $500,000 purchase price, 10-year useful life, 4.25% interest rate, $50 per hour operator labor cost.
§ Piece rate is the per-tray wage paid to workers and does not include indirect employment costs.

TABLE 5. Net income per acre for robotic harvest scenario number 2 (includes secondary pass of manual workers; piece rate for secondary pick assumed 150% of market rate)*†‡

Standard piece rate§	Robotic harvest efficiency					
	40%	50%	60%	70%	80%	90%
$1.70	−$3,129	−$1,337	$413	$1,799	$3,186	$4,572
$1.80	−$4,236	−$2,339	−$486	$982	$2,449	$3,917
$1.90	−$5,344	−$3,341	−$1,385	$164	$1,713	$3,263
$2.00	−$6,452	−$4,343	−$2,284	−$654	$977	$2,608
$2.10	−$7,559	−$5,346	−$3,184	−$1,471	$241	$1,954
$2.20	−$8,667	−$6,348	−$4,083	−$2,289	−$495	$1,299
$2.30	−$9,774	−$7,350	−$4,982	−$3,106	−$1,231	$645
$2.40	−$10,882	−$8,352	−$5,881	−$3,924	−$1,967	−$9
$2.50	−$11,989	−$9,354	−$6,780	−$4,742	−$2,703	−$664
$2.60	−$13,097	−$10,356	−$7,679	−$5,559	−$3,439	−$1,318
$2.70	−$14,205	−$11,359	−$8,579	−$6,377	−$4,175	−$1,973

* Shading indicates that the net return with a manual secondary harvest is higher than robotic harvest without secondary harvest.
† Assumes a 10-second per berry pick time.
‡ Machinery cost calculations based on Edwards (2019) and assume $500,000 purchase price, 10-year useful life, 4.25% interest rate, $50 per hour operator labor cost.
§ Piece rate is the per-tray wage paid to workers and does not include indirect employment costs.

TABLE 6. Net income per acre for robotic harvest scenario number 2 (includes secondary pass of manual workers; piece rate for secondary pick assumed 200% of market rate)*†‡

Standard piece rate§	Robotic harvest efficiency					
	40%	50%	60%	70%	80%	90%
$1.70	−$5,665	−$3,450	−$1,277	$531	$2,340	$4,149
$1.80	−$6,921	−$4,577	−$2,276	−$361	$1,554	$3,470
$1.90	−$8,178	−$5,703	−$3,275	−$1,253	$769	$2,790
$2.00	−$9,435	−$6,830	−$4,273	−$2,145	−$17	$2,111
$2.10	−$10,692	−$7,956	−$5,272	−$3,037	−$803	$1,432
$2.20	−$11,948	−$9,082	−$6,271	−$3,930	−$1,589	$753
$2.30	−$13,205	−$10,209	−$7,269	−$4,822	−$2,374	$73
$2.40	−$14,462	−$11,335	−$8,268	−$5,714	−$3,160	−$606
$2.50	−$15,719	−$12,462	−$9,266	−$6,606	−$3,946	−$1,285
$2.60	−$16,975	−$13,588	−$10,265	−$7,498	−$4,732	−$1,965
$2.70	−$18,232	−$14,715	−$11,264	−$8,390	−$5,517	−$2,644

* Shading indicates that the net return with a manual secondary harvest is higher than robotic harvest without secondary harvest.
† Assumes a 10-second per berry pick time.
‡ Machinery cost calculations based on Edwards (2019) and assume $500,000 purchase price, 10-year useful life, 4.25% interest rate, $50 per hour operator labor cost.
§ Piece rate is the per-tray wage paid to workers and does not include indirect employment costs.

harvest crew. As expected, this assumption reduces the range of wages and robotic harvest efficiencies for which a supplementary pick is more profitable, but there are no combinations in this table that indicate a net income that is higher than a single robotic harvest pass as well as the status quo manual harvest system.

Challenges and implications for the industry

The results show that at current market wage rates and harvest efficiencies presently attainable by leading robotic harvest technologies, manual harvest is more profitable than robotic harvest. Of course, this is not surprising, as there are not yet robotic harvest systems in widespread commercial operation in the United States. The results also show that if the robotic harvest systems can achieve efficiency rates above 70% or 80% of human harvest efficiency, and wage rates increase as expected over the next couple of years, the industry will see adoption of robotic harvesters become economically feasible for large strawberry growers, or as a custom harvest service.

This analysis relies heavily on assumptions of robotic harvest technical parameters, but also on machinery ownership and operating costs. Furthermore, there are several aspects of a strawberry production system that would be impacted by the adoption of a robotic harvest system that are not considered here. First, this analysis makes no mention of potential differences in fruit quality or marketability of robotically harvested fruit. There may be challenges associated with packaging fruit that have not been considered, and that will add cost to the robotic system. It's also possible that growers/shippers may be able to develop novel packaging solutions that will allow for more efficient distribution and sales under a robotic system. Given that there remains a role for human workers in all of the robotic harvest scenarios considered here, it appears that supply chains that can easily and efficiently accommodate both human and robotic harvesters will be necessary.

A second source of uncertainty relates to field cleanliness and plant health. Under the first robotic harvest scenario considered, no manual harvest is expected to follow the robotic harvest. The reasoning behind this assumption is that the high cost of a supplementary harvest will more than offset the additional revenue from the secondary pick and that yield lost to potential disease pressure caused by rotting fruit on the plant bed will be minimal. Future research will need to be carried out on the degree to which lower harvest efficiencies cause problems for overall plant health and reduce strawberry yields over the course of the season.

It is also unclear how the adoption of robotic harvest systems by some growers may impact the labor market for manual harvest crews on which the rest of the strawberry industry will continue to exclusively rely. If large producers are able to invest in robotic harvest systems first, thus reducing their demand for human workers, the growers that do not invest in robotic systems might see an easing of the labor challenges that they currently face. This would ultimately reduce the incentive for these growers to pursue automation and may lessen the rate of adoption. This analysis does not include modeling or estimates of the dynamics of adoption across the industry, other than speculate that larger growers are likely to be those that are incorporate the technology first.

Finally, a significant potential benefit of robotic harvest systems that is not considered in this analysis relates to the value of data that would be generated by near continuous plant-level monitoring with computer vision systems. High-resolution data on plant vigor, disease and insect pests could be gathered and applied in ways that are difficult to foresee today. It is beyond the scope of this work to estimate the benefit that this technology may ultimately yield, but high-resolution data has shown strong potential to profitably refine the management of other crops (Bauer et al. 2019; Trilles et al. 2018), and strawberry growers may be able to do the same.

Despite the technical challenges associated with the selective robotic harvest of sensitive ripe strawberries, industry leaders are optimistic that manual strawberry workers will eventually be supplemented by robotic harvest systems in some form. This analysis shows that while the field speeds and robotic harvest efficiencies are not yet to the point where these systems are competitive with human workers, a speed target of 10 seconds per berry and 70% to 80% harvest efficiency are not unreasonable goals and could make adoption of robotic harvest financially attractive in some form. Further, even with the commercial adoption of robotic harvest technology, this analysis shows that significant human labor will still be critical for fresh-market strawberry harvest in the United States. CA

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The potential threat of branched broomrape for California processing tomato: A review

After a 40-year absence, branched broomrape has reappeared in commercial California tomato fields, raising concern and prompting the search for integrated approaches to management.

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Online: https://doi.org/10.3733/ca.2021a0012

Abstract

Branched broomrape (*Phelipanche ramosa*), a parasitic weed that was the focus of a $1.5 million eradication effort four decades ago in California, has recently re-emerged in tomato fields in several Central Valley counties. Processing tomatoes are important to the California agricultural economy; the state produced over 90% of the 12 million tons of tomatoes grown in the United States in 2018. Branched broomrape is listed as an "A" noxious weed by the California Department of Food and Agriculture (CDFA); discovery of broomrape in California tomato fields leads to quarantine and crop destruction without harvest, resulting in significant economic loss to growers. In countries where broomrape is common, yield reductions caused by this parasitic weed can range from moderate to 80%, depending upon the infestation level, host and environmental conditions. Developing a detailed understanding of the biology of this weed under local conditions is an important step towards developing effective management plans for California. In this review, we discuss branched broomrape in the context of California production systems, particularly of tomato. We also discuss the potential management practices that could help to prevent or reduce the impacts of branched broomrape in tomatoes and other host crops.

Processing tomatoes are important to the California agricultural economy; in 2018, California accounted for over 90% of the 12 million tons of tomatoes grown in the United States (USDA NASS 2019). Some of the most potentially damaging pests of tomato include the weedy broomrapes (*Orobanche* and *Phelipanche* spp.), which have recently made an appearance in several California tomato fields after a 40-year hiatus. While broomrape is not currently at levels that can impact yield, presence in a field causes a large economic loss to growers because of the weed's status as a quarantine pest. The establishment and spread of broomrape in California tomato production regions could cause severe consequences for individual growers and the entire tomato industry.

Broomrapes are obligate root parasitic plants that can cause devastating damage to tomatoes and many other economically important broadleaf crops. These weeds use a modified root, called a haustorium, to fuse into a host plant root and extract nutrients and water. This greatly reduces productivity and sometimes kills the host. Globally, seven broomrape species have been identified that can cause damage to crops. Of

UC Davis graduate student researcher Matthew Fatino and Emeritus UC Cooperative Extension Farm Advisor Gene Miyao conduct early season scouting for branched broomrape in a field trial at a commercial processing tomato field site. *Photo:* Bradley Hanson.

these, small broomrape (*Orobanche minor*), Louisiana broomrape (*Orobanche ludoviciana*), Egyptian broomrape (*Phelipanche aegyptiaca*) and branched broomrape (*Phelipanche ramosa*) are known to be economically important pests in the United States (Jain and Foy 1989; Miyao 2016).

Tomato is highly susceptible to both branched broomrape and Egyptian broomrape. (A comparison of these broomrapes is shown in table 1.) Branched broomrape is currently classified in California as an "A" pest. An "A" pest is an organism of known economic importance subject to state-enforced action involving "eradication, quarantine regulation, containment, rejection, or other holding action" (CDFA 2020). The discovery of branched broomrape in a commercial tomato field leads to quarantine and crop destruction without harvest; processers will not accept a load of tomatoes from an infested field.

Egyptian broomrape, which, like branched broomrape, has been detected in some California tomato fields (Miyao 2016), is listed as a "Q" species. "Q" species have a temporary "A" classification pending determination of permanent rating by the state. Though Egyptian broomrape is currently considered less of a threat to California tomato crops than branched broomrape, Egyptian broomrape is also highly destructive. Studies in Israel showed that at high infestation levels (~100 shoots per square meter [m^2]), Egyptian broomrape can cause processing tomato yield losses as high as 70% (fig. 1). In Chile and Israel, annual economic losses in tomato due to Egyptian broomrapes have been estimated at $5 and $200 million, respectively (Hershenhorn et al. 2009).

Globally, branched broomrape is one of the most damaging and widespread of the weedy broomrape species, infesting nearly 6 million acres (about 2.6 million hectares) of broadleaf crops across Asia, the Mediterranean basin and North Africa (Mauromicale et al. 2008) (fig. 2). Branched broomrape infests a wide range of crops including tomato, cabbage, potato, eggplant, carrot, pepper, beans, celery, peanut and sunflower (table 2). A broomrape-parasitized plant suffers growth and yield reduction, and death can result in cases of severe infestation. Yield reduction can be significant depending on the level of infestation, susceptibility of the host and environmental conditions (Bernhard et al. 1998; Kogan 1994). Growers have reported up to 80% tomato crop loss due to branched broomrape in Chile (Kogan 1994). This is highly concerning given the similarity in tomato production systems and broomrape species with California.

The spread of branched broomrape

Branched broomrape was first documented in Europe in the 17th century (GBIF 2019), and is now present in 24 countries in Europe, North and South America, Africa and Asia (GBIF 2019; Mohamed et al. 2006). Most of the countries or locations where branched broomrape is reported have a Mediterranean climate, with warm-dry summers and rainy winters (fig. 2).

In the United States, branched broomrape was first reported in 1890 and, since then, over 150 occurrences have been documented (GBIF 2019; Musselman 1996). Reports of branched broomrape in the United States have been increasing, from seven occurrences in 2015 to 65 in 2019 (GBIF 2019), and it has been documented in Texas, Virginia, South Carolina, Illinois, New Jersey, Tennessee, Kentucky, Alabama and California (GBIF 2019; USDA-APHIS 2019). In California, branched broomrape was first seen in Butte County (1903) and later in Alameda County (1929) (Hrusa 2008).

TABLE 1. Comparing branched and Egyptian broomrape*

	Branched broomrape	Egyptian broomrape
Branching	Has branched stalk/shoot	Has branched stalk/shoot
Stalk height	Usually 15–20 cm tall, but can be up to 30 cm	Usually 20–30 cm tall or more
Peculiar morphology	No leaves and no green color on the whole plant	No leaves and no green color on the whole plant
Flowers	1. Flowers are merged with outer part appearing pale purple. 2. White cushions appear on lower lobe close to the base. 3. Flower tubes are widest at the top but narrowest at the base. 4. Length of flower is commonly less than 20 mm. 5. Anthers are sparsely hairy at the base.	1. Flowers are merged with outer part appearing pale blue or purple. 2. White cushions appear on lower lobe close to the base. 3. Flower tubes are widest at the top but narrowest at the base. 4. Length of flower is commonly longer than 20 mm. 5. Anthers are densely hairy.

* Molecular markers have been developed to distinguish between branched and Egyptian broomrape.

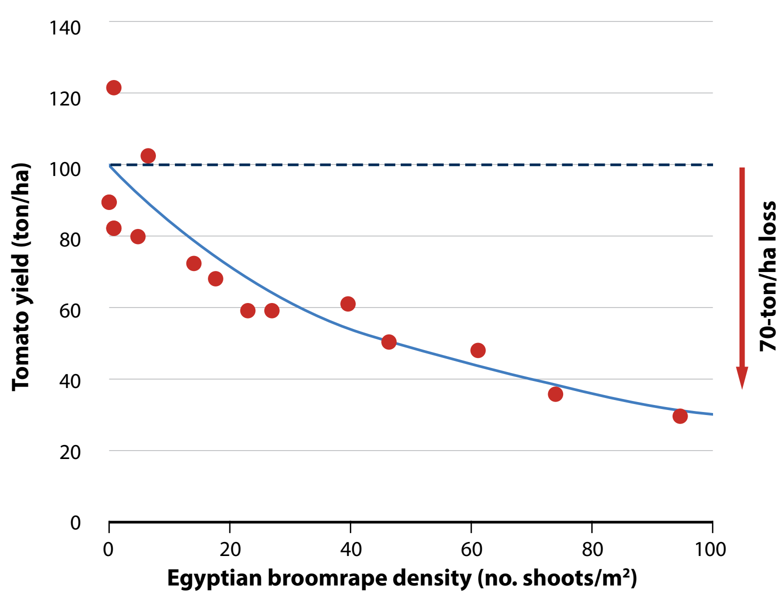

FIG. 1. Tomato yield loss caused by Egyptian broomrape density (H. Eizenberg, unpublished data; used with permission).

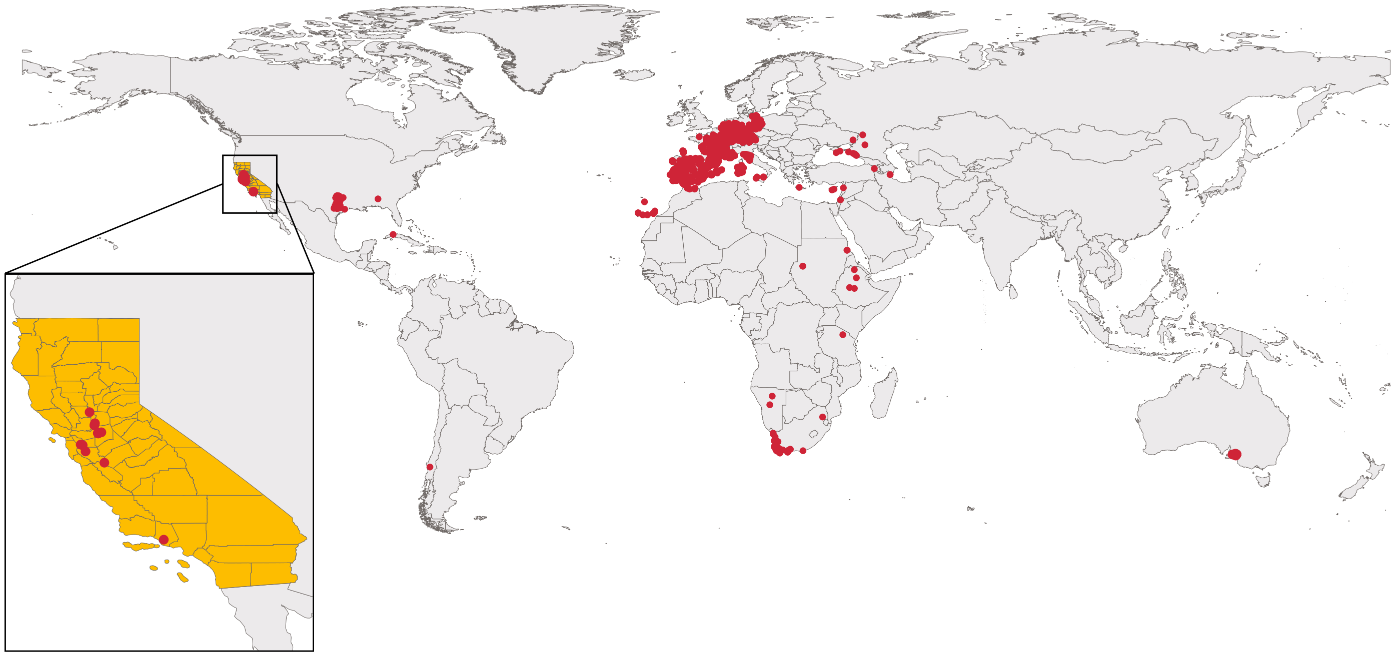

FIG. 2. Global distribution of branched broomrape. *Data source:* Calflora 2019, GBIF 2019 and ALA 2020.

TABLE 2. Host plants for branched broomrape relevant to California

Common name	Scientific name	Family	Reference
Cabbage	*Brassica oleracea*	Brassicaceae	Boari and Vurro 2004
Canola	*Brassica napus*	Brassicaceae	Benharrat et al. 2005
Carrot	*Daucus carota* L.	Apiaceae	Mauromicale et al. 2005
Celery	*Apium graveolens* L.	Apiaceae	Americanos 1991
Chickpea	*Cicer arietinum*	Fabaceae	Qasem and Foy 2007
Clovers	*Trifolium* spp.	Fabaceae	Amri et al. 2013
Eggplant	*Solanum melongena* L.	Solanaceae	Virtue et al. 2014
Faba bean	*Vicia faba*	Fabaceae	Sauerborn and Saxena 1986
Hemp	*Cannabis sativa* L.	Cannabaceae	Gonsior et al. 2004
Lentil	*Lens culinaris* Medik	Fabaceae	Buschmann et al. 2005
Lettuce	*Lactuca sativa*	Asteraceae	Panetta and Lawes 2007
Parsley	*Petroselinum crispum*	Apiaceae	Cochavi et al. 2015
Parsnip	*Pastinaca sativa*	Apiaceae	Kasasian 1971
Peanut	*Arachis hypogaea* L.	Fabaceae	Jain and Foy 1989
Pepper	*Capsicum fruitisence*	Solanaceae	Qasem 2009
Potato	*Solanum tuberosum* L.	Solanaceae	Haidar et al. 2003
Squash	*Cucurbita pepo*	Cucurbitaceae	Virtue et al. 2014
Sunflower	*Helianthus annuus* L.	Asteraceae	Karačić et al. 2010
Tobacco	*Nicotiana tabacum* L.	Solanaceae	Lolas 1994
Tomato	*Solanum lycopersicum* L.	Solanaceae	Mauromicale et al. 2008

Eventually it spread to other counties in California, including Colusa, Sacramento, San Benito, Santa Clara, San Joaquin, Ventura and Yolo (Calflora 2019; GBIF 2019; fig. 2).

A severe infestation of branched broomrape in the Sacramento Valley in 1959 prompted an intervention that involved soil fumigation with methyl bromide; this was as an industry-led effort funded through a legislative marketing order program (Jain and Foy 1989; Wilhelm 1965). The effort, which lasted from 1973 to 1982 and cost over $1.5 million (CTRI 2019), involved research, intensive field surveys and fumigation of infested fields and equipment to target the soil seedbank. As a result of those endeavors, branched broomrape became a less significant problem. Recently, however, this parasitic weed has been detected in tomato fields in Yolo, Solano and San Joaquin counties (Miyao 2016; figs. 2 and 3).

The cause of the re-emergence of branched broomrape remains unclear, although re-introduction or recurrence from long-dormant seed in the soil and subsequent spread are the most likely explanations. The re-emergence of this species in California is of concern to the processing tomato industry for many reasons: (1) the demonstrated global vulnerability of tomato to branched broomrape parasitism; (2) the similarity of California's climate to the species' native climate; (3) repeated cultivation of processing tomato in the same fields; (4) the cultivation of a wide range of hosts besides tomato (e.g., carrot, sunflower, safflower) in California; (5) intensive agricultural practices that could rapidly spread broomrape seeds to uninfested fields; (6) the plant's prolific production of tiny seeds that can easily disperse via machinery and irrigation water in the highly mechanized and irrigated cropping systems of California; (7) the ability of seeds to persist in the absence of hosts due to seed longevity of more than 20 years; (8) the difficulty of using conventional means of weed control, such as cultivation and contact herbicides, because so much of the plant's lifespan occurs underground; (9) the lack of some known important management tools (e.g., herbicides known to be effective in controlling broomrapes) because they are not yet registered or tested in California; and (10) regulatory and environmental challenges with soil fumigation practices.

FIG. 3. Branched broomrape can be difficult to detect in processing tomato fields due to its small stature. Its extended period of emergence and rapid progression from emergence to flowering (shown here) to having mature seed further complicate control strategies. *Photo:* Matthew Fatino.

Life cycle and physical characteristics

Branched broomrape is a holoparasite, meaning that it obtains all its nutrients from the host. Seed germination depends on the presence of a suitable host plant (Musselman 1996) and on prevailing environmental conditions. Seeds need to undergo a pre-conditioning period in the form of warm stratification before they can germinate (Fernández-Aparicio et al. 2016). The pre-conditioning period requires moist and warm (59°F [15°C] to 68°F [20°C]) environmental conditions from 5 to 21 days. The conditioned seed then can germinate in response to a signaling compound (strigolactone) released from the host plant root (Joel et al. 2007; fig. 4). If conditions remain conducive, multiple flushes of germination can occur within a single season (fig. 5); however, in the absence of stimulants, these preconditioned seeds re-enter dormancy. As the environment becomes drier, the seed's ability to germinate gradually reduces.

After germination, the radicle of the broomrape seedling grows a few millimeters in length and attaches to the host plant (fig. 4). If it fails to attach to a host

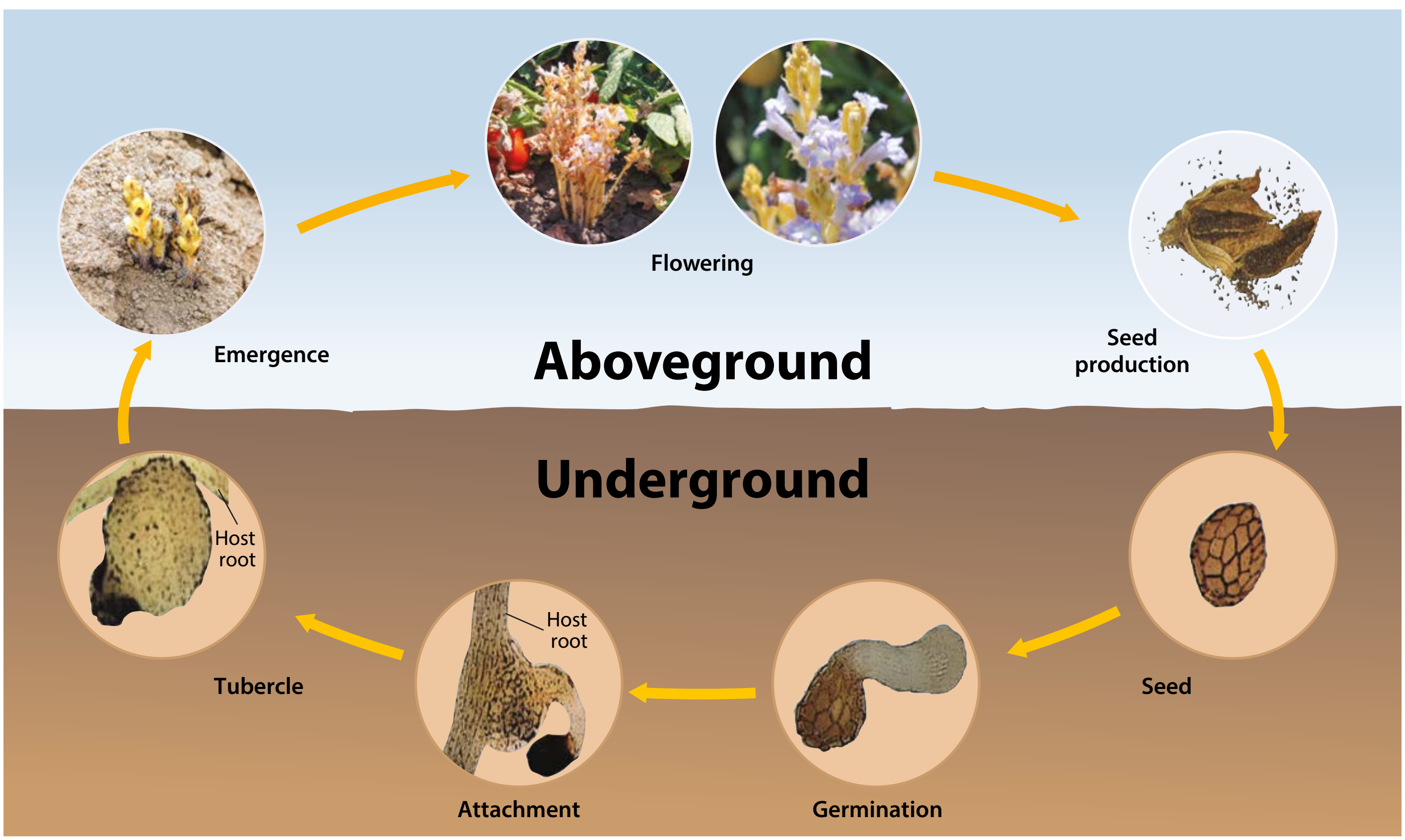

FIG. 4. A summarized life cycle of a branched broomrape. Modified from Eizenberg and Goldwasser (2018).

within a few days, the radicle exhausts its food reserves and dies (Fernández-Aparicio et al. 2016). Following attachment to the host plant, the radicle develops into a specialized modified root called a haustorium, a plant organ common to all parasitic plants (Buschmann et al. 2005). The haustorium fuses into the vascular system of the host root and serves as the bridge for extraction of nutrients and water from the transport systems (phloem and xylem) of the host (Fernández-Aparicio et al. 2016). Once connected to a host plant, broomrape grows rapidly, forming a tubercle — a storage organ for nutrients and water extracted from the host

FIG. 5. An infested tomato field with flags of different colors representing multiple flushes of branched broomrape captured weekly from May 29 to July 30, 2020, at Woodland, Yolo County, California. *Photo:* Matthew Fatino.

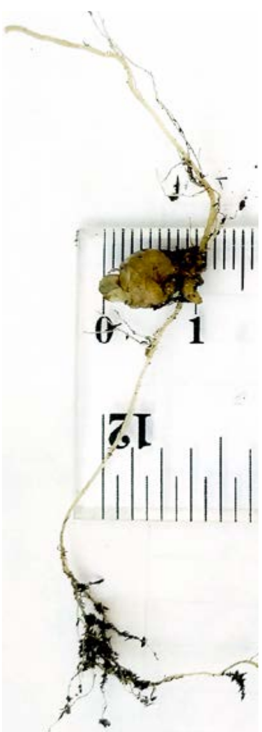

FIG. 6. A 1-cm-long tubercle of branched broomrape on a tomato root. *Photo:* O. Adewale Osipitan.

FIG. 7. Tubercle of a weedy broomrape with three shoots. *Photo:* O. Adewale Osipitan.

FIG. 8. A branched broomrape plant attached to a volunteer tomato root. *Photo:* Bradley D. Hanson.

— underground (figs. 6, 7 and 8). Multiple shoots develop from the tubercle and emerge above the soil surface, then grow to stalks from 6 inches (15 centimeter [cm]) to 12 inches (30 cm) in height (figs. 7 and 9). The shoots, wrapped with alternate bracts, completely lack leaves and chlorophyll. Prior to flowering, young plants look like yellowish spikes (fig. 9).

Flowering begins within 3 to 7 days after a broomrape shoot emerges above the soil surface (fig. 10). Branched broomrape flowers are spike-like, irregular, bisexual and usually pale white to purple in color. The petals of the flower are merged, tubular and have an upper and lower lip (fig. 10). The carpels are usually united to form a single chamber on the upper part of the flower; this chamber matures as a capsule with thousands of very tiny seeds, each smaller (0.2 millimeters to 0.4 millimeters) than a grain of sand (fig. 11). Seed production can occur within 14 days after flowering. A mature broomrape plant can produce hundreds of thousands of tan or brown-colored seeds, which can remain dormant and viable for many years (> 20) in soil. The entire life cycle, from seed germination to seed production, takes place within the March-to-August growing season of processing tomatoes in California.

Management: An integrated approach

Effective control of broomrapes is difficult, largely due to its unique biology and complex life cycle. As indicated above, most of the broomrape life cycle occurs below the soil surface, which makes it difficult to detect and control before it causes damage to the host plant. The short time period between emergence and seed dispersal also makes detection and control difficult, while the absence of chlorophyll and photosynthesis limits potential herbicide target sites and complicates chemical management. The tiny, hard-to-detect and abundant seeds, and the ability of the seeds to remain viable for decades, promotes the spread and persistence of branched broomrape in crop production systems. Thus, effective management of broomrape will require a long-term, integrated approach that involves sound understanding of the biology of the parasitic weed and the dissemination of information about management practices to all stakeholders.

Prevention and containment

Early detection and awareness of a new infestation, rapid reporting of the infestation to the local agricultural commissioner, proper removal of the branched broomrape plants, and management of the seedbank are crucial steps for successful containment and eradication of this parasite. Preventing the spread of branched broomrape is the most important component of the integrated approach to managing the weed. A current containment approach used in California is based on a quarantine regulation that places a recently

FIG. 9. Recently emerged broomrape shoots just starting to flower. *Photo:* O. Adewale Osipitan.

FIG. 10. A branched broomrape plant: flowering (left), maturing (center) and mature capsules (right). *Photos:* O. Adewale Osipitan.

FIG. 11. Tiny branched broomrape seeds (0.2–0.4 mm) and the single capsule from which the seeds were sourced. *Photo:* O. Adewale Osipitan.

[E]ffective management of broomrape will require a long-term, integrated approach that involves sound understanding of the biology of the parasitic weed and the dissemination of information about management practices to all stakeholders.

infested field on hold for a period of at least 2 years; in subsequent years, only rotational crops approved by the local agricultural commissioner may be cultivated in the field.

Upon detection of a new infestation, all branched broomrape plants should be removed carefully (e.g., pulled out of the soil by hand), ideally before they produce seeds. However, because of variability in the plant's growth stages (figs. 3 and 5), seed production might already have occurred by the time they are detected. The application of broad-spectrum herbicides at this stage, although likely to kill both the host plant and parasite, is less likely to affect the seeds. Therefore, the plants should be pulled and placed in plastic bags to minimize seed addition to the seedbank. The bags, tightly sealed, can be left under the sun (solarized) for a few days to promote the degradation of seeds. The plant materials (with or without solarization) can also be burned or destroyed by autoclave.

Weed seeds are often dispersed among fields by human activities, such as the transportation of contaminated farm produce (seeds, fruits and forage), the movement of contaminated vehicles and implements, and the spreading of contaminated soil and manure. Therefore, substantial effort should be made to clean and disinfect all equipment used in a field with broomrape infestation. Equipment sanitation should begin with removal of plant and soil debris manually, as debris not only can contain seeds but can also reduce the effectiveness of disinfectants. Once most of the debris has been removed, chemical disinfection agents can be used on the equipment to kill any remaining seed and pathogens. According to Hershenhorn et al. (2009), several quaternary ammonium products are available for disinfestation (phytosanitation) of farm equipment, such as didecyl dimethyl ammonium chloride, alkyl dimethyl benzyl ammonium chloride, dioctyl dimethyl ammonium chloride, octyl decyl dimethyl ammonium chloride, and ammonium bromide. Commercial products may contain one or a combination of these chemistries. For example, New Development Process (Process NPD; STERIS Corporation, St. Louis, Mo.) is an example of a commercially available product, containing multiple quaternary ammonia, that can be used for disinfection of farm equipment, clothing and shoes against broomrape seeds.

Soil fumigation using methyl bromide is one of the most effective tools to kill broomrape seeds, but due to its environmental toxicity the chemical has been banned and is no longer generally available. Even if methyl bromide were allowed under quarantine restrictions, the cost of treatment would be prohibitive. Other soil fumigants, such as chloropicrin, dazomet, metam-sodium, metam-potassium and 1,3-dichloropropene, may also provide different control levels of broomrape seeds (Eizenberg and Goldwasser 2018; Miyao 2016). However, at this time, few of these fumigants have been evaluated experimentally under California conditions.

Herbicidal control

Herbicidal control of broomrape can be undertaken using pre-plant and post-plant herbicide applications and/or chemigation (herbicide application through irrigation systems). This is an area of on-going research in California and builds on programs developed in other regions. In processing tomato in Israel, for example, herbicides have been used to effectively and economically manage broomrapes in highly infested fields where eradication is no longer feasible (Eizenberg and Goldwasser 2018). Growers found that pre-plant herbicide applications followed by complimentary post-transplant applications of acetolactate synthase-inhibiting herbicides such as sulfosulfuron (37.5 grams of active ingredient per hectare [g a.i. ha^{-1}]) provided control (~ 90%) of Egyptian broomrape at both pre- and post-attached stages in tomato (Eizenberg and Goldwasser 2018). The use of rimsulfuron (37.5 g a.i. ha^{-1}) as a pre-plant incorporated herbicide with a complimentary post-emergence application also provided good suppression (~ 70% control) of broomrape without causing significant damage to tomato plants (Eizenberg and Goldwasser 2018).

Some herbicide application protocols are based on the level of severity of broomrape infestation in tomato. For example, researchers in Israel have developed a thermal time-based decision support system (DSS) named PICKIT that takes into account infestation levels and growing degree days (GDD) since planting to guide the timing and rate of multiple herbicide applications for control of Egyptian broomrape; the system has been applied on a broad commercial scale (Eizenberg and Goldwasser 2018). For severe infestations (more than five broomrape plants per square meter), growers apply sulfosulfuron (37.5 g a.i. ha^{-1}) three times post-planting at 200, 400 and 600 GDD, followed by overhead irrigation (300 m^2) complemented by two foliar-applied doses of imazapic (4.8 g a.i. ha^{-1} each) at a later growth stage. The DSS suggests that a medium level of broomrape infestation (three to five plants per square meter) requires a single pre-plant incorporation of sulfosulfuron (37.5 g a.i. ha^{-1}) before planting tomato, followed by drip chemigation of imazapic (2.4 g a.i. ha^{-1}) at 400, 500, 600, 700 and 800 GDD, with two additional foliar imazapic applications (4.8 g a.i. ha^{-1}, each) at a later growth stage. A similar DSS system is being tested on branched broomrape infestations in processing tomatoes in Chile and California with promising initial results (fig. 5).

In California, only the rimsulfuron component of the PICKIT system is currently registered for use in processing tomato. Crop safety and registration-support research is ongoing in California in an effort to register additional herbicides and application techniques in the event that branched and/or Egyptian broomrape problems expand in scale (Fatino et al. 2019). A preliminary result from this research suggests that no visual injury and yield loss are associated with the use of the PICKIT system in local tomato fields (Fatino et al. 2019).

Cultural practices

Cultural practice, such as rotating tomato plants with false hosts (trap crops) or non-host crops, could help with seedbank depletion, provided branched broomrape seed is not re-introduced to the field from outside. A trap crop is a species with root exudates that induce broomrape seed germination but the crop does not allow attachment or support broomrape seedling growth and survival. Potential trap crops for branched broomrape that can be used in a rotation are alfalfa (*Medicago sativa*), cowpea (*Vigna unguiculata*), green pea (*Pisum sativum*) and flax (*Linum usitatissimum*) (table 3). Tomato and other host crops (table 2) should be excluded from the rotation for several years to encourage further depletion of seedbank with no chance of seed production. Since broomrape seed is very sensitive to flooding, incorporation of flooded rice into the crop rotation may also accelerate the depletion of soil seedbank (Goldwasser and Rodenburg 2013).

Soil fertility management can contribute substantially to the management of branched broomrape. Direct contact with fertilizer, such as urea and ammonium, may be toxic to broomrape, inhibiting seed germination and seedling growth (Fernandez-Aparicio et al. 2016; Westwood and Foy 1999). The negative effect of ammonium on broomrape is due to the plant's limited ability to detoxify the ammonium compound using glutamine synthetase (Fernandez-Aparicio et al. 2016). Application of adequate fertilizer will not only ensure unhindered growth of the tomato plant; it will also minimize the release of the plant's strigolactone, a root exudate that stimulates broomrape germination (Yoneyama et al. 2007). For example, it has been demonstrated that phosphate fertilization negatively impacts branched broomrape seed germination in tomato fields because of reductions in strigolactone exudation (López-Ráez et al. 2008).

Soil solarization has been shown to be an effective alternative to fumigation in reducing broomrape seed viability in areas with sufficiently hot climate. Solarization can significantly increase top soil temperatures up to 6 inches [15 cm] in depth when moist soil is covered with transparent polyethylene sheets for a period of one to two months. Dahlquist et al. (2007) reported 100% seed mortality of several weed species with solarization that raised soil temperature above 45°C for at least 96 cumulative hours. Mauro et al. (2005) found that soil solarization for two consecutive summers provided 99% mortality of viable seeds of branched broomrape in the seedbank without any negative impact on tomato yield. A recent field study conducted at UC Davis confirmed that soil solarization plus organic amends of either tomato pomace or plowed-down tomato plants can be used to substantially reduce the weed seedbank in general in tomato fields (Osipitan et al. 2020), although broomrape was not present at this site. One challenge in using this approach is the need to take tomato fields out of production for several months during the summer growing season in California. Additionally, it is not currently known if the elevated temperatures from solarization would penetrate deeply enough into the soil to provide adequate control of broomrape seed throughout the tomato root zone in an open-field production system.

Other thermal methods of soil disinfestation, such as soil steaming, are another alternative to chemical fumigation. Soil steaming (injecting low-pressure saturated steam into soil) has been shown to be effective in controlling seeds of several weeds and other soil pest in California strawberry production (Fennimore et al. 2014). High soil temperatures of 158°F (70°C) for 30 minutes can be regularly achieved in the field to a depth of 0 to 10 inches (25 cm) (Fennimore et al. 2014). This treatment seems to be sufficient to kill seeds of many weeds (Fennimore and Goodhue 2016; Melander and Kristensen 2011). Although the effect of this technique on broomrape seed mortality has not been studied, the small seed size of broomrape plants and their lack of protective tissues suggest that broomrape could be vulnerable to steam heating. However, like solarization, it is not known whether the depth of control from soil steaming would be sufficient as part of an eradication strategy for a quarantine pest like branched broomrape.

TABLE 3. Potential trap plants for branched broomrape

Common name	Scientific name	Family
Alfalfa	*Medicago sativa* L.	Fabaceae
Caraway	*Carum ajowan* Benth. et Hook	Apiaceae
Castor bean	*Ricinus communis* L.	Euphorbiaceae
Cowpea	*Vigna unguiculata* L.	Fabaceae
Flax	*Linum usitatissimum* L.	Fabaceae
Garlic	*Allium sativum* L.	Alliaceae
Green bean	*Phaseolus vulgaris* L.	Fabaceae
Green gram	*Vigna radiata* (L.) Wilczek	Fabaceae
Green pea	*Pisum sativum* L.	Fabaceae
Lablab bean	*Lablab purpureus* L.	Fabaceae
Ochrus pea	*Lathyrus ochrus* L.	Fabaceae
Sesame	*Sesamum indicum* L.	Pedaliaceae
Soybean	*Glycine max* L.	Fabaceae

Source: Kroschel 2002.

Physical control

Physical weed removal, such as hand weeding, particularly for a small infestation, can be part of an integrated approach to broomrape control. California is a state where hand removal of broomrape may be an option given the limited infestation level and widespread use of farm labor. The efficacy of hand weeding is highly dependent on thorough scouting and detection, which can be very difficult given the plant's small stature and the short period between its emergence and seed set (fig. 3). Deep inversion plowing (to more than 12 inches [30 cm]) would bury broomrape seeds to a depth below the soil layer where attachment to tomato root can occur (Eizenberg et al. 2007). However, the dormancy and durability of broomrape seed in the soil seedbank would increase the risk of later broomrape re-occurrences. Physical removal and deep burial could be part of a management strategy if broomrape became too widespread for quarantine and eradication efforts to be feasible; however, because broomrape is an A-listed pest (zero tolerance), physical removal and deep burial are not likely to provide a sufficient level of control alone.

Therefore, success will depend on significant funding from state or industry sources to offset grower costs in order to ensure grower participation and reporting.

Biological control

Biological control involves the use of biological agents or processes to damage seed, kill weedy plant or interfere with parasite-host relationships. A few examples of biological control of broomrapes have been reported in the literature. An insect herbivore, *Phytomyza orobanchia*, is known to be specific for broomrapes and feeds on broomrape ovules and seeds, thereby reducing broomrape seed production (Fernández-Aparicio et al. 2016). Pathogens such as *Fusarium* sp. (e.g., *Fusarium oxysporum* and *Fusarium arthrosporiodes*) can be incorporated into the soil to control broomrape through an induced cytoplasm metabolism and endosperm cell-wall degradation that breaks seed dormancy, thereby depleting the broomrape seedbank (Cohen et al. 2002). Pathogen-based herbicides have been reportedly used to control young seedlings of parasitic weeds (Abbasher and Sauerborn 1992), and these bioherbicides can provide complete control of all emerged broomrapes if formulated with multiple pathogens (Dor and Hershenhorn 2003; Müller-Stöver and Kroschel 2005). However, to date, no research on the applicability of these approaches in California cropping systems and broomrape infestation levels has been conducted, and they are not currently available for use.

Cultivation of resistant tomato varieties would also be an effective approach to prevent parasitic effects of broomrape. Resistance to branched broomrape might be achieved by incorporating traits that prevent haustorium attachment and penetration, or tubercle formation; this approach has been demonstrated in broomrape-resistant sunflower (Velasco et al. 2012). A group of scientists at UC Davis are currently screening a wide range of tomato varieties to determine their resistance to branched broomrape; results from this study could help to determine if enough genetic variability exists in tomato to use conventional breeding approaches to breed for broomrape resistance. Although screening is effective in small plots and is promising in the longer term, at present there are no effective commercial biological measures for broomrape control in tomato.

Conclusion

The re-emergence and spread of branched broomrape are of great concern in tomato and other susceptible crop production systems in California. At this point in time, the problem is still relatively small. Current efforts are focused on quarantine and eradication using a regulatory approach and soil fumigation. These approaches depend on the reporting of new infestations and generally result in total crop loss to the grower and extremely high treatment costs. Therefore, success will depend on significant funding from state or industry sources to offset grower costs in order to ensure grower participation and reporting. In the event that broomrape problems in California expand beyond what can realistically be managed using quarantine approaches, management and mitigation approaches will be needed just like with other widespread weeds. Other countries have successfully demonstrated that an integrated approach on a long-term basis, involving outreach to growers, field scouting and detection of new infestations, mapping of contaminated areas and fields, equipment sanitation, manipulation of cultural practices and carefully timed herbicide treatments, among other treatments, can effectively reduce yield losses caused by branched broomrape. Significant research efforts are being made by a group of university, industry and regulatory scientists to develop detection and management approaches for branched broomrape and to modify existing approaches from other regions for adaptation in California. CA

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RESEARCH ARTICLE

Point- and reach-scale measurements are important for determining accurate seepage rates in controlled flow channels

Point- and reach-scale field measurements are evaluated as a means to measure seepage rates in controlled flow channels at two locations in California's Central Valley.

by Mark E. Grismer

Online: https://doi.org/10.3733/ca.2021a0013

Abstract

A critical component of water-resources management in the irrigated agriculture landscape, particularly those landscapes dependent on groundwater availability, is determining groundwater recharge rates from streams and other channels. In California, flows in many such channels are "controlled" by upstream reservoir releases to meet downstream urban, irrigation and environmental water requirements. Seepage volumes from these channels and how they might vary during controlled release periods is a key component of meeting downstream riparian and groundwater-pumping needs. Understanding annual seepage from streamflow channels is also important in developing water budgets as part of the management of groundwater resources under the Sustainable Groundwater Management Act (SGMA) in California. However, direct measurements of channel seepage rates are infrequent or unavailable, and these rates, or associated volumes, are most often only estimated. Here we describe direct point- and reach-scale field measurements of channel seepage rates in Lower Putah Creek (Solano County) and in distribution lateral channels of the Oakdale Irrigation District on the east side of the San Joaquin Valley (San Joaquin and Stanislaus counties). We measured overall average seepage rates of about 2 feet (610 mm) per day at both locations and determined how these rates varied spatially and temporally during the summer when channel flows are controlled for downstream requirements.

In many semi-arid and arid regions, irrigated agriculture is the largest water user across the landscape. Irrigation has a profound impact on water allocations for other uses critical to civilization (e.g., urban supplies, industrial supplies and hydropower) as well as on local riparian environments (Lankford 2013). However, water diversions in earthen channels used in the irrigation process can result in substantial seepage to shallow groundwater storage. Seepage volumes vary depending in part on the water levels (flows) within streams or diversion channels and in part on field irrigation water-management; both are central to assessment of groundwater recharge rates. Several studies that employ soil-water balance and other methods have estimated rates of groundwater recharge for several irrigated crops and conditions in California (e.g., Grismer et al. 2000; Grismer 2012; Grismer and Asato 2012; Platts and Grismer 2014; Zikalala et al. 2019). Only recently have studies begun considering the effects of groundwater augmentation, particularly the flooding of cropped fields and orchards during winter rainy periods. These studies continue to rely on soil-water-balance methodologies even though they include some direct measurements of changes in shallow groundwater levels associated with recharge (e.g., Dahlke et al. 2018).

An irrigation canal in the San Joaquin Valley. Quantifying possible groundwater recharge from streams and earthen channels is important in developing water budgets under the Sustainable Groundwater Management Act. *Photo:* AJ Borba.

Groundwater recharge practices may become an important management tool in irrigated basins subject to periodic drought, adding some resiliency to the landscape hydrology. Though efforts have been made to identify likely streamflow-groundwater recharge areas across the state (Dahlke and Kocis 2018), estimations of seepage rates in the stream channels or canals in these areas are usually based on underlying soil texture. Few, if any, direct measurements of channel seepage are available for use in surface-to-groundwater modeling efforts required for developing local groundwater sustainability management plans under the Sustainable Groundwater Management Act (SGMA).

Similarly, a key element of integrated water resources management both locally and at watershed scales is an assessment of net seepage volumes to groundwater from canals, streams, lakes and the deep percolation (and associated lag times) from excess rain or irrigation, all of which enable quantification of a basin in water-balance terms. In most groundwater basins under consideration for management plans, such quantification has relied on modeling efforts where annual basin seepage quantities are determined as closure terms in the water balance. This is because direct measurement of seepage rates at the regional, or watershed, scale is problematic.

Typically, local seepage rates from ponds and canals in irrigated regions were measured directly (if at all) and, in some cases, deep percolation rates below irrigated lands were estimated from subsurface drainage flow rates (Grismer et al. 1988; Grismer and Tod 1991; Tod and Grismer 1991). Grismer and McCullough-Sanden (1988) and later Grismer and McCullough-Sanden (1989) determined seepage rates from subsurface drain-water collection ponds in the San Joaquin Valley using a combination of direct infiltrometer seepage measurements, lab-measured hydraulic conductivities of soil cores and water-balance calculations. They found that seepage rates from multiple infiltrometers set in the pond for several weeks were relatively small, approximately 0.04 to 0.4 inches (1 to 10 millimeters [mm]) per day, and that the seepage was log-normally distributed spatially in the clay loam soils comprising the base of most ponds. Somewhat surprisingly, the log-normal mean of the hydraulic conductivity of the matching soil cores taken from the infiltrometers was six to seven times smaller than the average seepage rates measured using the infiltrometers and determined from water-balance calculations. In Iraq, Mohsen and Mohammed (2016) also used direct infiltrometer measurements, finding canal seepage losses of roughly 1.3 feet (0.4 meters [m]) per day in lined distribution channels and 13.1 feet (4.0 m) per day in unlined channels.

Sonnichsen (1993) summarized historical results of canal seepage studies in the United States and created figure 1, which illustrates the log-normal and grain-size dependence of measured seepage rates from lined and unlined canals. Consistent with the infiltrometer results above, seepage rates in these studies ranged from about 0.09 feet (30 mm) per day for concrete-lined canals to about 1.9 feet (600 mm) per day for canals having sandy-gravel bottoms. In all these studies, however, seepage rates estimated from grain-size distributions had greater variabilities and so were of limited value in predicting actual seepage rates. Such results indicate the complexity of estimating seepage volumes to groundwater based on soil-textural information. Moreover, these studies determined seepage rates only in the top roughly one foot of bed materials. Though it was assumed that seepage eventually reached shallow groundwater within the next 10 to 300 feet (3 to 100 m)

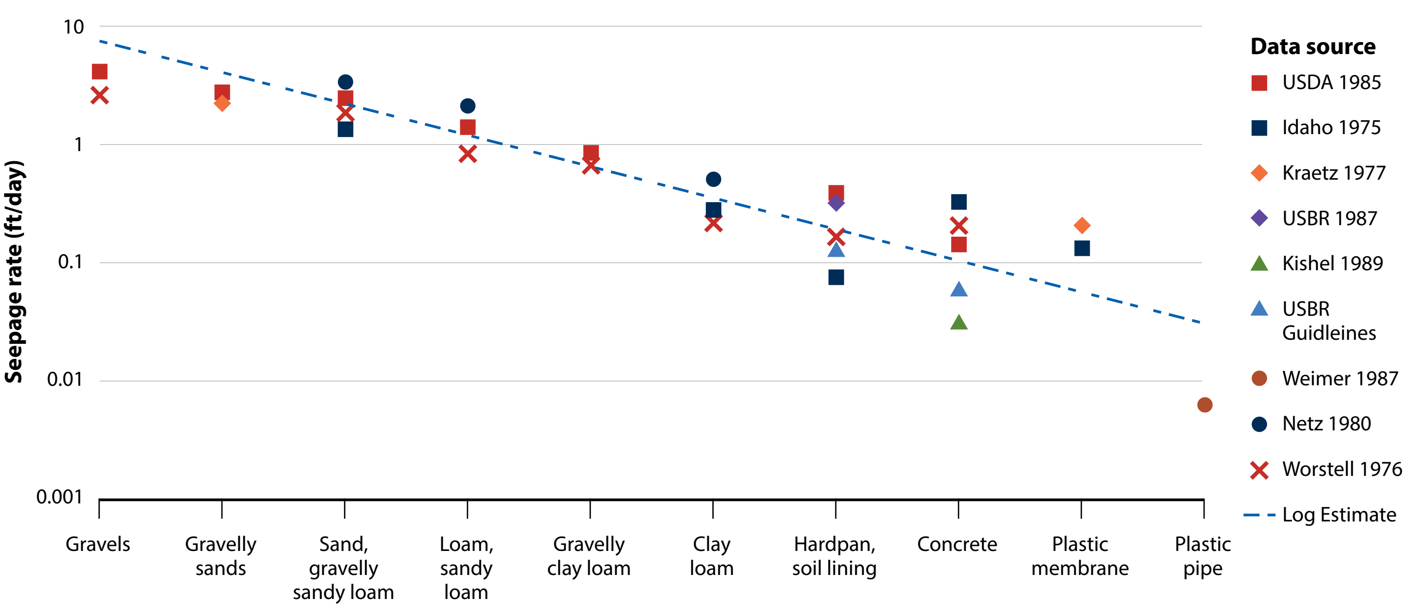
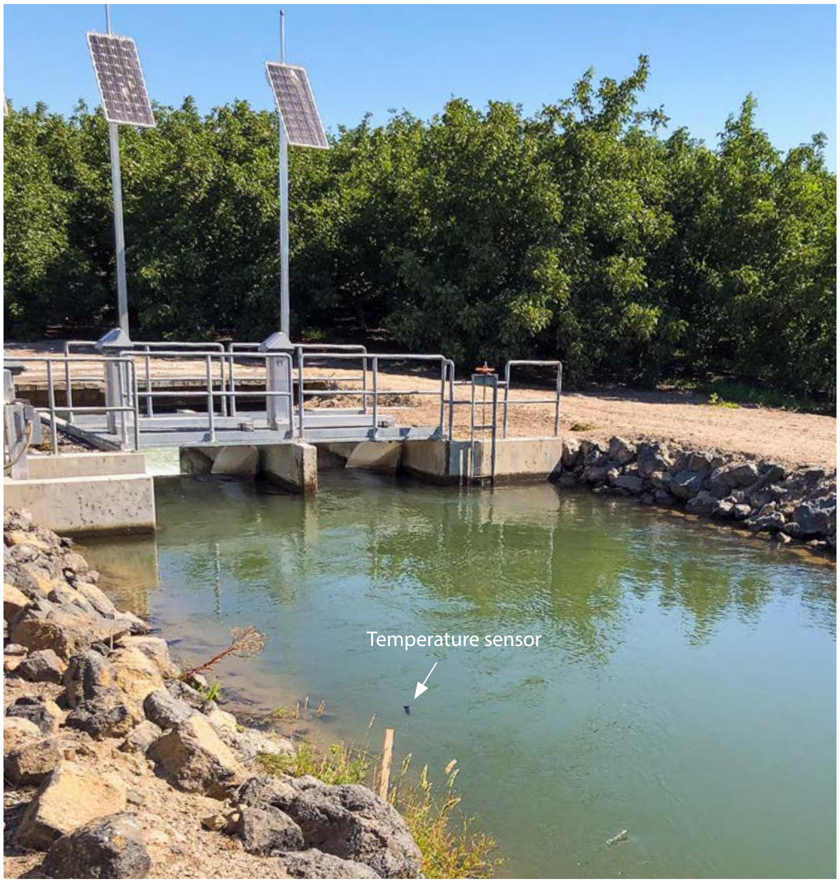


FIG. 1. Historically measured canal seepage rates from unlined canals and typical linings in the United States (from Sonnichsen 1993). Canal seepage rates decrease by orders of magnitude with decreasing grain size of bed materials, or replacement by liners.



A water temperature sensor visible in the water, near an automated gate (background). *Photo:* AJ Borba.

Aerial image of OID Claribel reach, with approximate location of automatic gates (CLB-D3, CLB-D4) and WL sensor (CLB-004-001). *Photo:* AJ Borba.

depths, it is possible that much seepage was distributed laterally, augmenting stream flows or appearing as downslope seeps.

Aside from the studies noted above, few contemporary direct measurements of seepage rates or volumes exist in the literature, and this remains a large gap in understanding the irrigated hydrological landscape as well as in developing hydrology management plans like those associated with SGMA in California.

Using a combination of field-measurement and water-balance calculations, we measured channel seepage in two locations in California: at the point scale for Lower Putah Creek (LPC) in Solano County and at reach scales in several delivery laterals of the Oakdale Irrigation District (OID) in Stanislaus and San Joaquin counties. We also examined the effects of channel water levels (flowrates) on measured seepage rates to gain insight into how manipulation of channel flows may change channel groundwater recharge rates. In both cases, quantifying possible groundwater recharge from the channels is important in SGMA related groundwater planning in both Solano and Stanislaus counties.

Site descriptions and methodology

Both project sites are in California's Central Valley, one (LPC) on the west side near the Sacramento-San Joaquin River delta, the other (OID) on the east side at the western edge of the Sierra Nevada foothills. At both sites, channel reaches — sections of the stream or river — pass through agricultural regions of moderate relief that have similar topography and similar elevations, from near sea level to roughly 300 feet (100 m) above sea level. The sites also share similar Mediterranean climates, including daily temperatures and evapotranspiration rates, though the LPC area receives slightly more average annual rainfall (approximately 24 inches [610 mm] per year compared to roughly 21 inches per year at Oakdale). Prevailing soil textures at both sites ranged from silty to sandy-gravel loam, though at the LPC infiltrometer sites we encountered relatively dense clay layers within the top several inches below the streambed, likely resulting from extensive gravel mining within the creek over the past several decades. Shallow groundwater levels or water tables in both regions during the summer months are 10 to 100 feet (3 to 30 m) below ground surface but rise during winter rainy periods and spring flooding to near-surface levels.

The LPC serves as the boundary between Yolo and Solano counties, while the OID straddles the Stanislaus River, which supplies the district canals within Stanislaus and San Joaquin counties. Solano County Water Agency (SCWA) controls summer flows into the roughly 12.4 miles (20 kilometers [km]) of the LPC that originate west of Winters at the Lake Solano Putah Diversion Dam and continue toward Davis to meet downstream user and contractual requirements as well as to sustain some riparian habitat. At perhaps a larger scale, the OID, one of the older water districts in the state, manages over 300 miles (500 km) of channels across a roughly 88,000-acre (35,300-hectare [ha]) service area surrounding the town of Oakdale. About 90% of the OID service area is located in Stanislaus County.

Lower Putah Creek

Due to limited access across private properties and the inability to restrict Putah Creek flows as required for reach-scale measurements, we used infiltrometers to obtain point seepage rates from several locations along the low-flow creek channel, from west of Winters to the Russell Ranch area west of Davis. We conducted streambed infiltration rate studies and associated computed hydraulic conductivity (K) measurements at four sites along the LPC; two were located roughly at 3 miles (5 km) upstream (west) of Winters, at Winters Putah Creek Park, and two were roughly 5 to 6 miles (8 to 10 km) east of Winters.

The four infiltrometers were constructed as described by Grismer and McCullough-Sanden (1988) from 5-foot (1.5-m) lengths of 6-inch (150-mm) diameter Schedule 40 (white) PVC pipe. We beveled the pipe edges of one end of each infiltrometer for ease of sediment penetration and outfitted each with an external clear nylon tube for direct measurement of water levels (WLs) within the pipe during infiltration tests (fig. 2). We carefully pushed the infiltrometers into the streambed sediment to an average depth of 0.5 feet ± 1.3 feet (0.17 m ± 0.4 m) at each location. Adjacent streamwater depths ranged from 0.25 to 1.5 feet (0.1 to 0.45 m) and water temperatures during the tests averaged 61°F (16°C) at the most westerly site and about 64°F (18°C) at all other sites further downstream. Air temperatures ranged from 70°F to 100°F (20°C to 38°C) during the infiltrometer tests, and evaporation from the infiltrometers was assumed negligible during the duration of each test (less than 60 minutes).

After we installed the infiltrometers, we filled them with creek water to about 3 feet (1 m) over the stream water level and allowed them to infiltrate for 15 to 30 minutes. We then refilled them, recorded the initial full-water level and then measured declining water levels within the columns every few minutes until water levels fell about 3 feet (1 m), or after about 60 minutes had elapsed. Replicated infiltrometer tests at five locations revealed that rates of water-level decline were nearly the same as those of the initial test. With water temperatures near 68°F (20°C), we made no temperature corrections on viscosity or density when computing the K values.

We analyzed the infiltrometer WL fall-rate data to determine K values, solving the Darcy equation for transient (falling-head permeameter) or quasi–steady state conditions. We assumed that the infiltrometer-enclosed sediment was part of a much longer column of saturated homogeneous soil. The transient-flow Darcy equation is formulated as a simple first-order rate equation that, when integrated over the time-period $\triangle t = t_1 - t_0$ when the infiltrometer WL falls from H_0 to H_t, results in a natural-log solution for the hydraulic conductivity (K_v), designated here as

$$K_v = [L/(t_t - t_0)]\ln(H_0/H_t),$$

where L is the sediment thickness, H_0 is the initial WL in the infiltrometer at time zero, t_0, and H_t is the infiltrometer WL at time t_t. This transient solution suggests that there is an exponentially decreasing rate of WL decline within the infiltrometer during the measurements. The steady-state solution of the Darcy equation for K_s assumes that the total head, H_t, on the top of the sediment core of length, L, contained within the infiltrometer is given by the infiltrometer WL, while H_s, representing the lower-boundary condition total head, is the stream water level. These boundary conditions applied as long as the enclosed core remained saturated and as long as the local water table was located at least several times the core thickness, L, below the base of the infiltrometer. Computing the average flux, q_{avg}, as the average rate of WL decline in the infiltrometer during the test, we determined the hydraulic conductivity as

$$K_s = q_{avg}/[(H_t - H_s)/L].$$

Oak Irrigation District

The OID installed automated-gate (WL) control structures along several key distribution laterals within the OID canals in 2019. We took advantage of these structures to develop reach-scale water balances to determine reach seepage rates within three channel sections located north to south across the district. The automated structures were designed to maintain nearly constant upstream WLs during the irrigation season (May to October), so we were able to obtain direct, instantaneous measurements of flow rates and reach WLs in each section. The control gates adjusted flows depending on delivery requirements, from one-minute intervals to roughly 60-minute intervals. This enabled us to compute seepage rates for each time interval. We determined seepage rates, W (L/T; e.g., inches per day

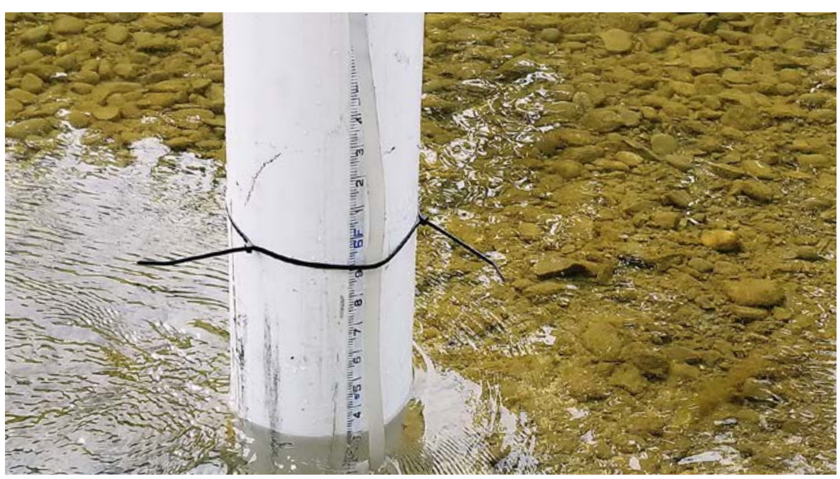


FIG. 2. Infiltrometer installed in a typical cobble/gravel Lower Putah Creek streambed under the shallow flow conditions found at most sites.

[in/day]), for each interval each day for each channel reach from a simple mass balance (with appropriate unit conversions) given by

$$W = [(Q_{in} - Q_{out} - Q_{div} - \text{Evap}*A_s)\triangle t - \triangle\text{WL}*A_s]/(A_U*\triangle t),$$

where

Q_{in} = inflow rate to channel reach (L^3/T; e.g., cubic feet per second [cfs]),

Q_{out} = outflow rate from channel reach (L^3/T, e.g., cfs),

Q_{div} = diversion flow rate from channel reach (L^3/T; e.g., cfs),

Evap = Rohwer (1931) equation or CIMIS evaporation rate (L/T; e.g., in/day)

A_s = reach surface area (L^2; e.g., square feet [ft^2]),

A_U = water surface area of unlined reach section (L^2; e.g., ft^2),

$\triangle$WL = change in reach water level during time $\triangle t$ (L, e.g., inches), and

$\triangle t$ = time interval between measurements of flowrates and WLs (T; e.g., minutes).

TABLE 1. Summary of Lower Putah Creek (LPC) infiltrometer test locations (west to east) and results

River mile	Northerly	Westerly	Sediment thickness, *L* (in)	Average gradient *△H/L* (in/in)	K_s (in/day)	K_v (in/day)
22	38° 30.119	121° 59.400	5.20	4.71	12.60	7.08
22	38° 30.119	121° 59.400	6.10	4.01	73.20	59.40
22	38° 30.119	121° 59.400	4.13	4.98	102.00	70.10
22	38° 30.133	121° 59.390	7.64	3.79	10.50	6.30
22	38° 30.133	121° 59.390	5.75	3.76	131.00	87.80
22	38° 30.141	121° 59.382	11.22	2.90	0.63	0.39
22	38° 30.141	121° 59.382	8.74	2.95	3.11	1.97
20	38° 31.215	121° 58.058	7.20	5.09	0.51	0.20
20	38° 31.215	121° 58.058	5.63	7.32	0.20	0.12
20	38° 31.235	121° 58.023	5.75	6.12	0.55	0.31
20	38° 31.327	121° 57.871	3.11	13.10	0.00	0.00
20	38° 31.239	121° 58.022	4.45	7.46	28.40	26.40
15	38° 31.818	121° 54.397	6.34	2.21	270.00	151.00
15	38° 31.818	121° 54.397	6.73	3.61	24.00	14.40
15	38° 31.818	121° 54.397	6.85	3.19	77.80	90.30
15	38° 31.818	121° 54.397	6.34*	2.31	312.00	161.00
15	38° 31.818	121° 54.397	6.73*	3.88	24.60	16.70
15	38° 31.818	121° 54.397	6.85*	3.36	74.80	43.50
15	38° 31.842	121° 54.342	6.22	4.77	71.60	72.80
15	38° 31.842	121° 54.342	6.10	6.23	7.56	8.11
15	38° 31.842	121° 54.342	5.24	7.98	0.98	0.63
15	38° 31.842	121° 54.342	5.94	2.65	486.00	395.00
14	38° 32.257	121° 52.094	5.71	6.44	17.90	16.40
14	38° 32.257	121° 52.094	8.62	3.99	25.30	21.30
14	38° 32.257	121° 52.094	9.25	3.41	125.00	118.00
14	38° 32.257	121° 52.094	9.25	3.72	122.00	96.40
14	38° 32.257	121° 52.094	7.95	3.97	60.60	46.40
14	38° 32.257	121° 52.094	7.13	4.67	36.80	36.00
14	38° 32.257	121° 52.094	5.98	3.34	440.00	504.00
14	38° 32.257	121° 52.094	5.98	3.77	453.00	420.00

* Replicate test.

We assumed in this water-balance equation that the water-surface area, A_U, was equal to the vertical projection of the channel soil-surface area as the channels had nearly rectangular cross-sections. That is, we effectively assumed that there was only vertical infiltration from the channel, or, lateral infiltration rates were insignificant. In the OID channel reaches that we studied, the channel side-slopes, while not vertical, were fairly steep, often greater than 45 degrees (1:1 horizontal to vertical), such that using the water-surface area A_U underpredicts the actual soil surface area by more than 7%. We deployed thermocouples with dataloggers to collect water temperatures every 30 minutes at each end of two reaches in the one lateral canal (Cometa) on the north side of the district and one lateral canal (Claribel) on the south side to estimate reach evaporation rates in conjunction with weather data (collected at the centrally located CIMIS micro-meteorological station in Oakdale). We determined reach water-surface areas from as-built canal survey drawings, and we assumed that the areas remained constant, as WLs varied by less than 0.4 inches (10 mm) within time intervals $\triangle t$, and by no more than 2.4 inches (60 mm) overall from day-to-day. We calculated seepage rates for each time interval $\triangle t$, and, following the analysis outlined below, averaged the seepage rates associated with time intervals $\triangle t$ of 10 minutes and greater to determine the daily seepage rate each day from June 1 to mid-October 2019.

Analyzing point seepage measurements at LPC

We outline the LPC infiltrometer test results first (table 1), and then consider the reach-scale seepage measurements at OID, before discussing how the results relate to possible groundwater recharge. The infiltrometer tests were relatively straightforward in the field, with the only practical issue being the presence at some locations of a shallow, hard clay layer within a few inches of the sediment surface, which made it difficult to drive the infiltrometers into the streambed. In nearly all tests, the infiltrometer WL (H_t) fell at a steady rate, as illustrated by the results from a set of three tests at the site furthest downstream (fig. 3). This steady rate of H_t decline in the infiltrometers as the test progressed was equivalent to a steady infiltration, or seepage rate, regardless of the declining gradient across the sediment core. This observation is inconsistent with the fundamental assumptions associated with the steady-state and transient solutions of the Darcy equation above.

The steady infiltrometer WL decline suggests that small changes in hydraulic gradient had little effect on measured seepage rates and that the steady-state solution is likely a better fit to the measured data to determine *K* than the falling-head solution. Nonetheless, for comparison purposes, values of K_s as well as those for the falling-head test, K_v, are listed in table 1. Across all 30 tests, the arithmetic average *K* was a relatively small value of about 8 feet (2.5 m) per day, but values ranged across five orders of magnitude, from <0.001 to 36 feet (11 m) per day, with a log-normal mean of ~2 feet (~660 mm) per day. Landon et al. (2001) obtained similar seepage results for several sandy-bottom creeks in Nebraska with log-normal means on the order of 4 inches (100 mm) per day.

The test hydraulic gradients, $\triangle H/L$, ranged from 3 to 7, with an average of 4.7, a value within the range of what might occur under regular summer flow conditions. Reflecting the log-normal distribution, about one-third of the tests yielded *K* values <0.3 feet (<0.10 m) per day, the next third between 0.3 and 3.2 feet (0.10 to 1.0 m) per day, and the remaining third between 3.2 and 36 feet (1.0 to 11 m) per day. Replicated tests (repeated immediately following the first trial) yielded *K* values similar to those of the initial value, with computed K_s values. Overall, K_s values averaged about 15% more than K_v values. Because the measured infiltrometer data were inconsistent with the required assumptions for calculation of K_v values, the following discussion focuses on the K_s values.

The variation in K_s values was largely dependent on location along the river, while variation found in test results at each site was less than the overall variation except for that at the Winters Park site (river mile [RM] 20) at the town of Winters. Streambed sediments captured within the infiltrometers at all sites were comprised of sands, silts, gravel and clay, and all sites had some cover of smooth cobble 1 to 3 inches (2 to 6 cm) in diameter (e.g., fig. 2), though there was much less surface gravel at the uppermost river site. The defining sediment feature with respect to the infiltrometer test results was the capture of dense clay at depths less than about 6 inches (152 mm) in several cases within the infiltrometer core. Table 2 summarizes the infiltrometer test-result averages for each of the LPC sites. While mean *K* values were greater at the two downstream sites (RMs 15 and 14) than those at RM 22, they did not differ at a significant level (>95%). Mean gradients and sediment thicknesses at these three sites also did not differ significantly (>95%). However, at the Winters Park site (RM 20), the average gradient and mean *K* value were significantly greater and smaller, respectively. These significant differences at the Winters Park site were directly associated with the dense, sticky clay found immediately below the riverbed sand/gravel mix.

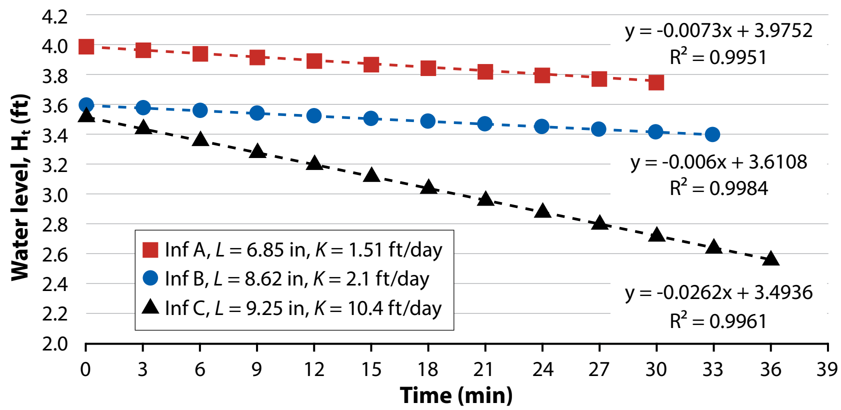


FIG. 3. Example infiltrometer test results from location RM 14 in Lower Putah Creek illustrating reproducibility of the method. Note the constant infiltration rate (line slope) with declining infiltrometer water level (H_t) in contrast to the log decay expected theoretically.

TABLE 2. Infiltrometer test-result averages for each Lower Putah Creek (LPC) site

River mile	Sediment thickness, *L (in)*	Average $\triangle H/L$ *(in/in)*	Average K_s *(m/day)*	Average K_s *(in/day)*
22	6.97	3.87	1.21	47.60
20	5.24	7.82	0.15	5.91
15	6.34	4.02	3.43	135.00
14	7.48	4.22	3.01	119.00

Relative to values measured at other, similar streams across the country, the average *K* within the range 3 to 10 feet (1 to 3 m) per day is relatively small, but not unreasonable. At the Winters Park and other river sites subject to historical in-channel mining, from the town of Winters upstream, the shallow, dense clay layer severely reduces channel permeability, suggesting that at these locations there may be very limited channel seepage. Such limited seepage upstream of Winters may have frustrated groundwater recharge efforts associated with increased streamflows during the recent drought period that failed to support riparian trees (Grismer 2018). However, we determined streambed material permeability only to the top 4 to 9 inches (10 to 20 cm) of sediment. Additional clay layers may be present at greater depths along the river that limit seepage to shallow groundwater tables during the summer when local water tables fall far below the channel low-flow water level. Reach-scale measurements of seepage are likely required to further evaluate the streamflow-groundwater interactions along the LPC riparian area.

Analyzing reach seepage measurements at OID

While reach-scale seepage measurements were not immediately possible at LPC, we were able to complete three reach-scale measurements of channel seepage in similar-size channels in the OID. We conducted this work along two key water distribution laterals south (Claribel) and north (Cometa) of the town of Oakdale. Both distribution laterals are operated with downstream WL control using the automated gates

TABLE 3. Oakdale Irrigation District (OID) canal reach locations and overall dimensions

Reach	Location	Central coordinates	Length (*km or feet*)	Surface area, A_s (*ha or ac*)	Surface area, A_u (*ha or ac*)
Claribel D3-4 (one turnout)	5.4 km south of central Oakdale	37°43′22.23″ N 120°49′24.67″ W	1.08 km or 3,548 ft	0.75 ha or 1.86 ac	0.73 ha or 1.78 ac
Cometa D1-2 (no turnouts)	6.3 km northeast of central Oakdale	37°49′15.64″ N 120°49′48.22″ W	1.05 km or 3,440 ft	1.15 ha or 2.85 ac	0.91 ha or 2.24 ac
Cometa D4-5 (one turnout)	7.6 km north of central Oakdale	37°50′05.00″ N 120°51′08.07″ W	2.18 km or 7,137 ft	2.28 ha or 5.63 ac	2.28 ha or 5.54 ac

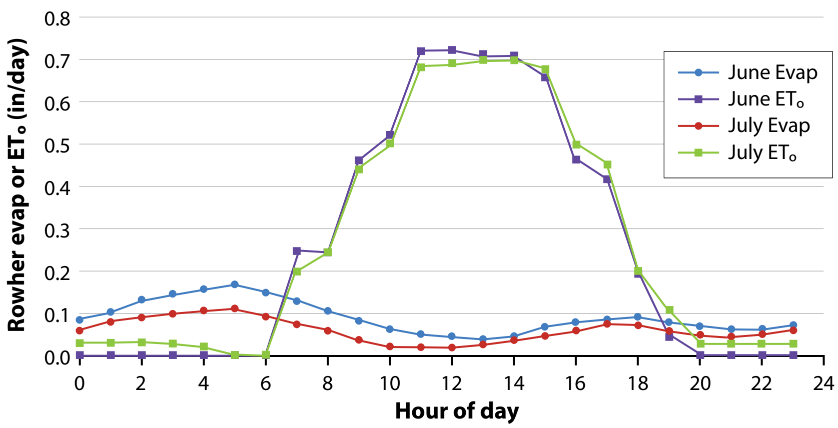


FIG. 4. Estimated hourly average evaporation rates during June and July 2019 for the Claribel canal reach illustrating substantial differences between ET_o and Rohwer equation approaches to calculation.

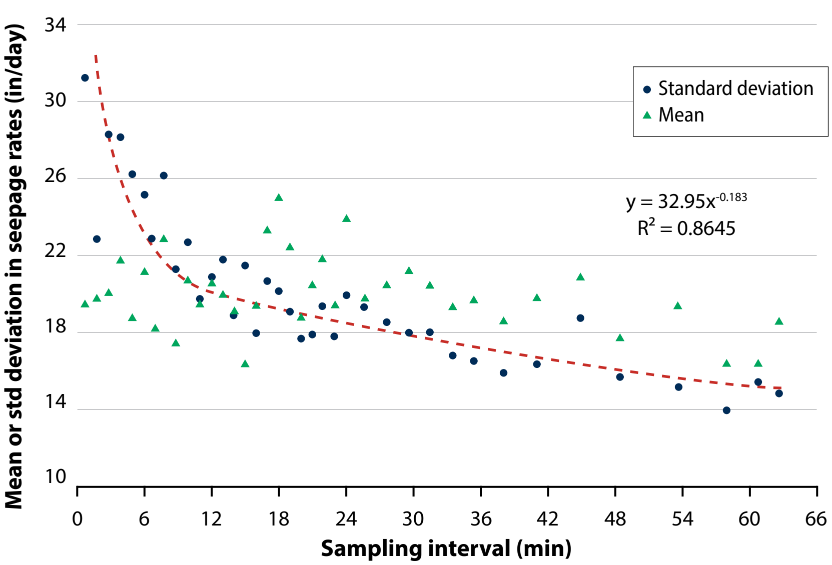


FIG. 5. Mean seepage rates and their variability during summer 2019 for the Claribel canal reach depend on water-balance sampling interval time. Higher deviations and slightly decreasing infiltration rates with increased sampling time suggest that a sampling interval of >20 min is appropriate for this reach.

mentioned earlier in place of the traditional manually adjusted weir board drop structures. The automated gates have WL sensors and gate position encoders that enable continuous flow measurement, real-time monitoring, automated control and data acquisition through a telemetry system. The reaches we selected within these automated distribution laterals were roughly 3,300 to 6,500 feet (1 to 2 km) long and 2.5 to 5 acres (1 to 2 ha) in surface area. The Claribel and Cometa D3-4 canal reaches had one diversion, or turnout, within the reach, and there were no diversions within the Cometa D1-2 reach. Flowrates across the reaches ranged from 20 to 150 cfs (0.6 to 4.3 cubic meters per second [m^3/s]) during the four-month monitoring period. Table 3 summarizes the reach locations and characteristics. We made roughly 100 seepage determinations from reach water balances every day at time intervals of 10 to nearly 70 minutes between changes in gate flows and slight changes in reach WLs. We anticipated that the estimated seepage rates from the more frequent (>1,000) and smaller time-interval water-balance calculations each day would be more variable than those from longer periods, and we assumed that reach evaporation would have minimal effect on computed seepage rates. We tested both concepts first using the daily data from Claribel, the smallest of the reaches considered here.

We calculated hourly evaporation rates for the water-balance determinations of seepage rates using both the Rohwer equation and from the unadjusted reference evapotranspiration (ET_o) measured at the Oakdale CIMIS station. The Rohwer equation for water-surface evaporation includes wind speed, humidity and the water-vapor pressure difference between that of the directly measured reach water temperature and that measured in the air (CIMIS station). We found that daily calculated water evaporation in the Claribel reach was much smaller than daily CIMIS values and that peak calculated evaporation rates during June and July surprisingly occurred in the early morning hours as compared to peak ET_o rates, which occurred at midday (fig. 4). Ultimately, while we included evaporation losses in our calculations, net evaporation volumes during the sampling time intervals of 10 to 60 minutes had negligible effect on the estimated seepage rates for the Claribel reach.

We anticipated decreasing variability in estimated seepage rates as sampling intervals (Δt) increased, and we were uncertain as to how this would affect mean seepage estimates. Based on all the data from the Claribel channel reach, we found that evaporation losses had negligible effects on mean seepage rates for each time interval. There was no change in seepage estimates when using the Rohwer evaporation rates, and there was an occasional reduction in seepage rates by ~1% using the CIMIS ET_o values, though we questioned their applicability given the observations shown in figure 4.

The effects of sampling interval, Δt, on mean seepage rates and their variability in the Claribel canal reach are shown in figure 5; there was a slight but nonsignificant decreasing trend in the mean seepage rates, but variability clearly decreased with increasing Δt. For sampling intervals greater than about 10 minutes, there was little significant decrease in variability and mean seepage rates; therefore, in all further analyses, we focused on determining daily seepage rates for sampling intervals greater than 10 minutes. We found similar trends between sampling interval and seepage variation for the other two OID reaches considered. Time intervals of 10 to 20 minutes also corresponded roughly with the average travel time required for water to flow across the length of the reach during the June-to-October period.

Mean daily seepage rates from the Claribel reach varied more-or-less randomly during the four-month measurement period, and we found similar decreasing trends at the Cometa reaches, particularly after mid-September (day 100) (fig. 6). Decreasing seepage rates were associated with decreasing average daily flowrates and reach WLs later in the season. For example, at Cometa D1-2, the mean seepage rate during the first 100 days after June 1, of 0.71 inches (18 mm) per hour, fell to 0.23 inches (5.9 mm) per hour for the last 42 days of record. This decrease was associated with declining water depth and associated declining average daily flow rates, which fell from 116 cfs (3.28 m^3/s) to 63 cfs (1.78 m^3/s) after 100 days. Table 4 summarizes the mean and standard deviations of the daily mean sampling intervals, counts and seepage rates from each canal reach. The overall daily means at all three reaches differed significantly at the 99% confidence level.

Average seepage rates across all three OID channel sections during the 100-day period encompassing June through mid-September were 2 feet (605 m) per day, or about 1% less than the log-normal mean seepage rate from LPC, which was 2.16 feet (660 mm) per day.

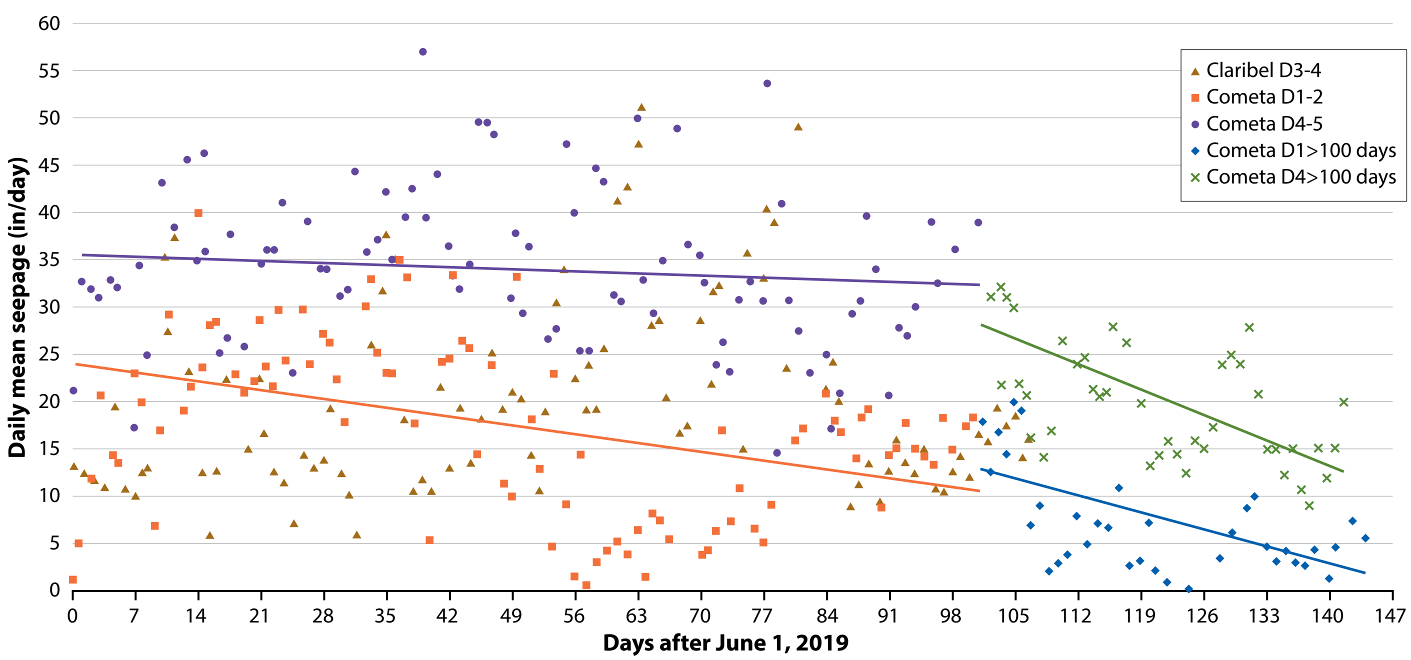


FIG. 6. Mean daily seepage rates at all three Oakdale Irrigation District (OID) sites show progressively decreasing values during summer 2019, especially during the early fall period when canal flowrates (water levels) decrease.

TABLE 4. Summary of mean daily values for each Oakdale Irrigation District (OID) canal reach

Canal reach	Count	Statistic	Δt *(min)*	Daily count	Seepage *(in/day)*	Seepage *(mm/day)*
Claribel D3-4	107	Mean	29.50	40.60	20.40	518
		Std. dev.	4.20	9.05	9.82	250
Cometa D1-2	144	Mean	20.79	42.40	12.90	329
		Std. dev.	3.47	5.80	9.82	250
Cometa D4-5	142	Mean	19.16	33.00	29.70	754
		Std. dev.	3.65	8.97	9.92	252

The Cometa and Claribel laterals are just a small part of the roughly 200-mile (322-km) lateral canal system below the main canals within OID. However, a total of 28 miles (45 km) of these smaller lateral canals now have automated downstream WL control systems. We assume that standard operations with the traditional weir board drop structures provide less head and opportunity time for groundwater recharge because the reaches are only checked when water deliveries are occurring in the reach immediately upstream.

OID is undertaking a phasing-in expansion of automated gate controls on other distribution laterals but does not anticipate implementation of such controls on smaller canals. Currently, there is a combined total unlined surface area of approximately 52 acres (25 ha) in automated control reaches across the district. Assuming that the average Cometa and Claribel reach seepage rate (2 feet [605 mm] per day) is representative of all other gate-controlled lateral canals, the possible groundwater recharge from the system is equivalent to about 123 acres per foot per day (15 hectare-meter [ha-m] per day). Thus, the combined seepage volume for the 100-day summer period accounts for more than one-third of the estimated annual seepage from OID's distribution system (over 12,000 acres per foot) and is likely a valuable component of the regional groundwater balance, particularly in the northern part of the district nearest to the over-drafted Eastern San Joaquin Groundwater Subbasin. With more detailed assessment of the LPC channel geometry and geology, the point-measured seepage rates could be similarly used to determine groundwater recharge volumes to the Solano and Yolo groundwater basins during the summer low-flow period.

Summary and conclusions

Quantification of seepage from channels subject to controlled flows from upstream reservoirs is an important but challenging aspect of water-resources planning across the irrigated agricultural landscapes common in California. By measuring deep percolation or groundwater recharge rates/volumes from such channels and determining their dependence on the managed flowrates, water resources planners will be better able to allocate limited water supplies to downstream needs, including riparian forests, irrigation and the augmentation of groundwater storage. Lacking direct measurements of channel seepage rates, planners and watershed modelers rely on soil-texture–implied seepage rates or those determined from closures of regional water balances. The in-channel infiltrometer point measurements we made along the LPC revealed the large spatial variability in likely seepage rates as estimated from *K* values and suggest that such measurements are too fine-scale to determine channel reach–seepage volumes. However, results from those measurements provided insight into that variability and suggested that typical stream WL changes have very limited effects on locally measured seepage rates. At the reach scale, the water-balance estimates of seepage rates and their changes during the summer at OID yielded direct assessment of likely groundwater recharge volumes important in water-resources planning associated with SGMA requirements in the region. Coincidentally, overall average seepage rates at both sites were similar — about 2 feet (610 mm) per day — across a range of soil textures. CA

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RESEARCH ARTICLE

Integration of grazing and herbicide application improves management of barb goatgrass and medusahead in pasture and rangelands

The combination of high-intensity, short-duration grazing with precisely timed applications of glyphosate improves management of invasive annual grasses.

by Travis M. Bean, Josh S. Davy, Guy B. Kyser and Elise S. Gornish

Online: https://doi.org/10.3733/ca.2021a0011

Abstract

The invasive annual grasses barb goatgrass (*Aegilops triuncialis* L.) and medusahead (*Elymus caput-medusae* L.) are widespread in western states and present management challenges on grasslands. To develop an integrated management strategy for these species, we treated sites in five pastures in Mendocino County, comparing combinations of intensive sheep grazing, glyphosate herbicide (low and high), and application timings (tillering, boot and heading stage). We found that grazing alone reduced barb goatgrass spikelet densities by 68% and the number of seeds per spikelet by 35%. Both rates of glyphosate application without grazing had similar effects on seed production. High and low glyphosate application at tillering resulted in almost complete control of both target species. Boot- and heading-stage applications reduced barb goatgrass density by 39% and 32%, respectively. Application at the boot stage also resulted in an 82% reduction in number of seeds per barb goatgrass spikelet. Our results suggest that intensive grazing may be a useful management strategy to reduce barb goatgrass and medusahead spikelet densities and barb goatgrass seed numbers, especially when integrated with a boot- or heading-stage glyphosate application.

The rapid spread of invasive annual grasses across noncrop and rangeland areas causes extensive economic and ecological damage across California and other western states. Dense infestations of weedy grasses can increase fire frequency and magnitude (D'Antonio and Vitousek 1992; Lambert et al. 2010a; Lambert et al. 2010b), modify virus incidence in native bunchgrasses (Malmstrom et al. 2005), impact soil microbial communities and nutrient cycling (Batten et al. 2006; Drenovsky and Batten 2007; Gornish et al. 2020), reduce native plant diversity (DiTomaso 2000; Haferkamp et al. 2001; Parmenter and MacMahon 1983) and reduce livestock carrying capacity (Hironaka 1961; Jacobsen 1929). In western habitats with Mediterranean climates, two invasive annual Eurasian grasses of particular concern are barb goatgrass (*Aegilops triuncialis* L.) and medusahead (*Elymus caput-medusae* L., syn. *Taeniatherum caput-medusae* [L.] Nevski). Barb goatgrass is listed as a noxious weed by California and Oregon, and is considered an invasive pest in Nevada and several New England states. Medusahead is a state-listed noxious weed in California, Colorado, Nevada, Oregon and Utah. Although research on medusahead has been more extensive than research on barb goatgrass, it remains one of the most problematic grasses in the western United States (James et al. 2015).

Barb goatgrass and medusahead are challanging invasive grasses to manage. However, new research suggests that an integrated pest management (IPM) approach using low rate herbicide application early in the plants' growth and grazing can be very effective for control.

Elena Zhukova

Failure of nonintegrated approaches

Both barb goatgrass and medusahead were introduced to the United States in the late 19th to early 20th centuries, and both have increased their ranges while responding poorly to nonintegrated management approaches (Peters 1994), especially over the long term (James et al. 2015). Both species typically invade areas dominated by functionally similar annual grasses, causing strong negative effects on the abundance of the similar species (Case et al. 2016).

Until recently, most research on controlling invasive annual grasses has focused on a single type of management (biological, chemical, cultural or mechanical) and has not included the effects of treatment timing. For example, prescribed burning can control barb goatgrass (DiTomaso et al. 2006) but, if burning is conducted too early in the plant's life cycle, temperatures or exposure time may not be sufficient to sterilize seed. If conducted too late, prescribed burning makes the seed less susceptible to injury (Sweet et al. 2008). High stocking rates of sheep at the onset of heading can reduce medusahead in small plots (DiTomaso et al. 2008) but, on a pasture scale, livestock tend to avoid injurious seed appendages, limiting practical applications to vegetative (tillering) and early reproductive (boot) stages. Mowing at the early heading stage, like grazing, can control both barb goatgrass (Aigner and Woerly 2011) and medusahead (Zhang et al. 2010), but rough terrain can limit equipment access. Chemical control can be effective on barb goatgrass (Aigner and Woerly 2011), but none of the herbicides currently registered for use on rangelands without grazing restrictions provides selective control without damaging desirable grasses (Peters et al. 1996). In addition, while the ideal timing of application is known for these control efforts independently, the most appropriate timing for combined approaches is less well understood. This is because the effects of a control effort at one instance during the life cycle of a weed can directly and indirectly affect the way the plant responds to a control effort at another point in its life cycle (Blumenthal et al. 2003; Melander 1998). Clearly, a better understanding of how combined control approaches interact with phenology and treatment timing would facilitate an improved strategy for barb goatgrass and medusahead management.

Medusahead.

Barb goatgrass.

Integrating management approaches

The main objective of this research was to compare the effectiveness of different application timings (at different developmental stages) using label- and reduced-rate glyphosate applications in combination with targeted grazing in reducing barb goatgrass and medusahead spikelet density and seed production. We hoped also to refine the herbicide application timing to allow use of a reduced rate. We looked at seed characteristics rather than cover because these species, like other invasive annuals, tend to dominate California landscapes in large part due to their copious seed production — not their enhanced competitive abilities (Seabloom et al. 2003).

We utilized a replicated field experiment on active rangeland with herbicide applications made at conspicuous target plant developmental stages: tillering (when side shoots are produced), boot (when inflorescence develops within the shoot) and heading (when inflorescence emerges from the shoot). We paid particular attention to the relevance and ease of use of the treatment procedures for land managers. To our knowledge, this is the first study to combine high-intensity, short-duration grazing of barb goatgrass and medusahead with precisely timed applications of different amounts of glyphosate, with a focus on reduction in seed production.

Establishing experimental sites

In fall 2015, we established plots in five pastures (table 1) at the University of California Hopland Research and Extension Center (HREC; headquarters coordinates 39.001774, ⁻123.084377). The Niderost, Little Buck and James pastures are productive lowlands adjacent to seasonal flow, while the Foster and South pastures are dry and open with some serpentine soils.

The HREC comprises nearly 5,400 acres (2,185 hectares [ha]) of grassland, oak woodland and irrigated pasture in the interior Coast Ranges of California. The climate at HREC is Mediterranean with hot, dry summers and mild, rainy winters. Annual precipitation averages around 40 inches (102 cm), with 75% of the precipitation received from November through February. Mean average temperature from July through September is 70°F (21°C), and the mean maximum is 92°F (33°C). July is generally the hottest month, with daily maximum

TABLE 1. Site locations and characteristics

Site	Coordinates	Elevation *(m)*	Soil
Niderost	38.987° N, 123.090° W	175–185	Yorkville-Squawrock-Witherell complex (loam, sandy loam and cobbly loam)
Little Buck	38.995° N, 123.068° W	280–290	Bearwallow-Hellman loam
James	39.031° N, 123.095° W	465	Talmage gravelly sandy loam
Foster	39.005° N, 123.101° W	265–275	Henneke-Montara complex (loam and gravelly clay loam)
South pasture	38.985° N, 123.066° W	250–270	Henneke-Montara complex (loam and gravelly clay loam)

sometimes reaching 110°F (43°C). Temperatures drop to a mean of 44°F to 47°F (7°C to 8°C) from December through February. The grasslands at HREC are generally on moderate slopes with loam to clay soils.

In each of the five pastures, or sites (Foster, James, Little Buck, Niderost and South), we established three replicated treatment blocks in random locations. Each block was 59 feet (ft) by 118 ft (18 meters [m] by 36 m) and utilized a split-split-plot design. A 59-ft-by-59-ft (18 m by 18 m) fenced grazing enclosure on the center formed the main-plot treatment. Glyphosate rate (split-plot treatments) and application timing (split-split-plot treatments) were established both inside and outside the grazing enclosure (fig. 1). Grazing treatments were applied in a standardized manner (spatially) across sites and blocks — not randomly — due to management constraints at the site.

Herbicide treatments

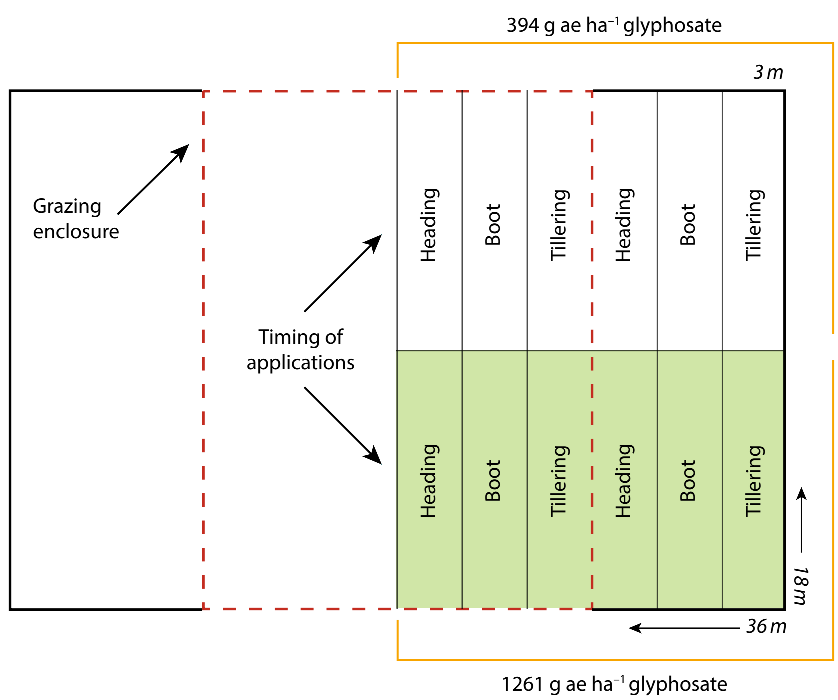
Glyphosate (Monsanto Roundup WeatherMAX 4.5 pounds [lbs] acid equivalent [ae] per gallon [gal^{-1}] [0.54 kg ae L^{-1}]) was applied to split plots at two rates: a low rate of 0.35 lb ae per acre (ac^{-1}) or 10 ounces (oz) product ac^{-1} (0.39 kg ae ha^{-1} or 0.3 L product ha^{-1}) and a high rate (label recommended rate for similar annual grasses) of 1.12 lbs ae ac^{-1} or 32 oz product ac^{-1} (1.26 kg ae ha^{-1} or 1.0 L product ha^{-1}). Application timings were targeted to one of three phenological stages: tillering, boot or heading. All applications were made in a spray volume of 20 gal ac^{-1} (187 L ha^{-1}) using a CO_2 backpack sprayer and a 10-ft (3-m) boom with six TeeJet 11002AIXR nozzles on 20-in (0.5-m) spacings.

Our categories represented continuous transitions from vegetative growth to flowering and senescence, not discrete categories, so slight differences in phenology existed among sites, as noted here. Tillering treatments were applied on March 22, 2016, when both barb goatgrass and medusahead were in a vegetative stage at all sites. This was also prior to flowering of other annual grasses. Boot-stage treatments were applied on May 9; at this time, while barb goatgrass at the Niderost, Little Buck and James sites were in the boot stage, plants at the drier Foster and South pasture sites had advanced to early heading. Other annual grasses had finished flowering and were beginning to senesce by this time. Heading-stage treatments were applied on May 27. At this time, barb goatgrass was fully headed out at all sites; spikelets were still green at James, spikelet awns were starting to redden at Little Buck and Niderost and spikelets were starting to brown at Foster and South pastures. A heading-stage treatment was not applied at South pasture due to equipment problems.

Sheep grazing treatments

Based on sheep-grazing rates of 10 animal days per 1,076-ft^2 plot applied to manage medusahead in DiTomaso et al. (2008), we planned a target rate of 32 sheep days in each of our 3,488-ft^2 plots (32 sheep for one day or 16 sheep for two days, approximately 0.2 animal unit month [AUM]). Plots were grazed at the boot stage prior to the boot-stage herbicide applications. The South pasture plots were grazed April 18–21; Foster, April 21–26; Niderost, April 25–27; Little Buck, April 28–30; and James, May 3–5. Because forage was denser

FIG. 1. An example of the layout of experimental treatments in a single block.



Reduced medusahead densities in grazed areas (left) compared to ungrazed areas (right).

at Niderost, Little Buck and James, sheep remained at these sites for an extra day (a total of approximately 48 sheep days) to achieve forage reduction consistent with forage reduction at South pasture and Foster.

Evaluations: weed spikelet density

Spikelet densities of both barb goatgrass and medusahead were evaluated on June 16. Three quadrats were tossed in each glyphosate-treated split plot, and six quadrats were tossed in the larger split plots not treated with glyphosate. We used 0.5-m^2 quadrats to evaluate split plots with high densities of barb goatgrass and 1-m^2 quadrats for lower-density split plots.

Evaluations: seed production

On the same date, June 16, we also collected 10 barb goatgrass spikelets from each split plot with mature plants. Most barb goatgrass plants in grazed split plots and in split plots treated with glyphosate at tillering were immature, if present; therefore, we returned to collect spikelets from these plots on July 21. At this time, we found almost no barb goatgrass plants in any of the plots. Medusahead density was too low across plots to allow for spikelet collection.

Assessing the models

We used ANOVA to assess how our treatments independently and interactively affected barb goatgrass and medusahead spikelet density, modeling both species separately. Our models included the fixed factors of grazing (absent and present); glyphosate rate (none, low and high); target species stage at glyphosate application (tiller, boot and heading), and the random effect of split plot within block within site. When significant effects were found, we used Tukey HSD tests to compare treatment levels. Barb goatgrass and medusahead spikelet density values were log transformed to accommodate the assumptions of ANOVA. We used a separate ANOVA model with the same fixed and random factors to look at treatment effects on log-transformed barb goatgrass seed number. Effects of the unbalanced design (due to equipment problems at the South pasture during one treatment) on ANOVA outcomes were investigated comparing results using the grand mean and the weighted mean (Algina and Swaminathan 2011). No differences were found so data were pooled across sites. All analyses were conducted in R version 3.2.2 (R Development Core Team 2008).

Barb goatgrass spikelet density

Grazing alone reduced overall barb goatgrass spikelet density by 68% (F = 43.44, $p < 0.001$) compared to ungrazed treatments. Glyphosate application alone reduced spikelet density by 60% overall (F = 99.61, $p < 0.001$, fig. 2), and effects were similar between low- and high-rate glyphosate treatment plots. Barb

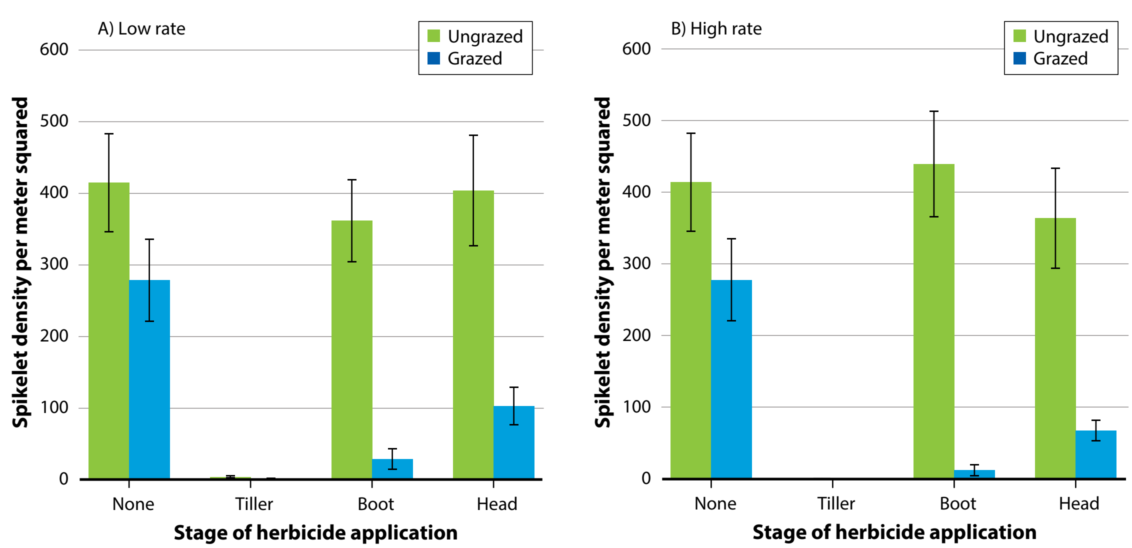


FIG. 2. Mean density ±standard error (SE) of barb goatgrass spikelets (m^{-2}) for grazed and ungrazed treatments combined with low (A) and high (B) rates of glyphosate applied at tiller, boot and heading stages.

goatgrass spikelet density was also affected by the stage at which the plants were sprayed (F = 190.82, $p < 0.001$). Spikelet density reductions were near 100% in the tillering stage, 39% in the boot stage and 32% in the heading stage when compared to the control. We found no interactions among grazing and glyphosate rates or among glyphosate rates and stage of herbicide application.

Medusahead spikelet density

Although two of the sites were chosen based on the presence of medusahead thatch from previous years, medusahead spikelet density was very low across experimental plots during the survey period (with no differences among plots). Out of all experimental treatments, only tiller-stage glyphosate application contributed to significant differences in spikelet density (F = 15.84, $p < 0.001$, fig. 3). Application of glyphosate to medusahead in the tiller stage resulted in a 99% decline in spikelet density compared to spraying at other stages. Spraying medusahead at the boot stage resulted in a 47% decline in spikelet density compared to spraying at the heading stage.

Barb goatgrass seed number

In general, barb goatgrass seed production was higher in the absence of grazing (mean = 4.2 seeds per spikelet, df = 2, F = 68.68, $p < 0.001$; fig. 4) than it was in the presence of grazing (mean = 3.1 seeds per spikelet). As well, barb goatgrass seed numbers showed a significant response to herbicide treatment at all stages of development. Seed numbers were higher in control plots not treated with glyphosate (mean = 5.6 seeds per spikelet; df = 2, F = 93.15, $p < 0.001$) compared to the plots treated with low and high rates of glyphosate (mean = 3.1 seeds per spikelet for both treatment levels, fig. 4A and 4B). Seed numbers declined with the interactive effect of grazing with glyphosate rate and stage of application (df = 2, F = 6.91, $p = 0.009$). When glyphosate was applied at the heading stage with grazing present,

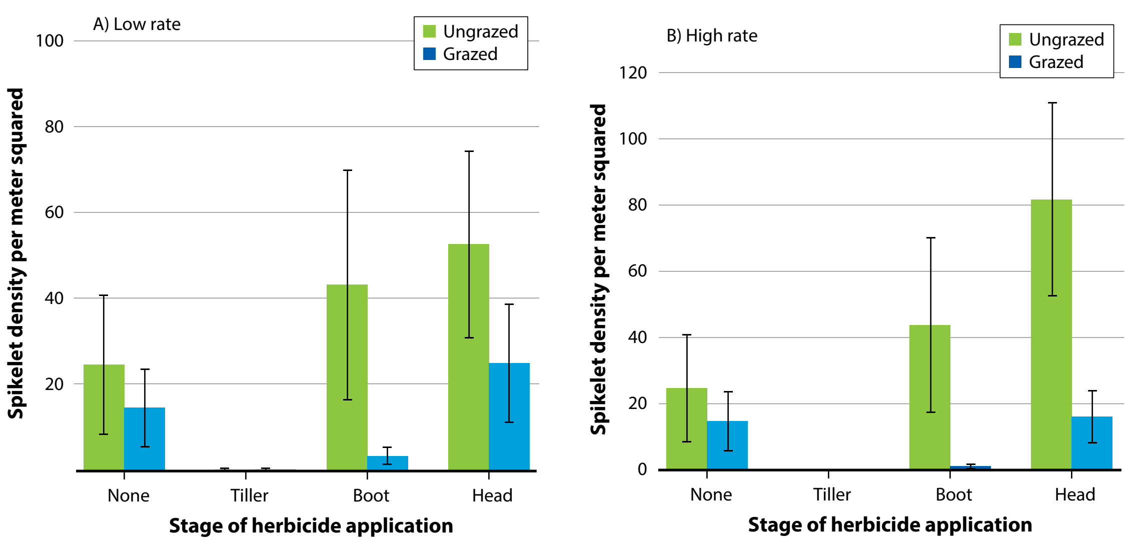


FIG. 3. Mean density ±SE of medusahead spikelets (m^{-2}) for grazed and ungrazed treatments combined with low (A) and high (B) rates of glyphosate applied at tiller, boot and heading stages.

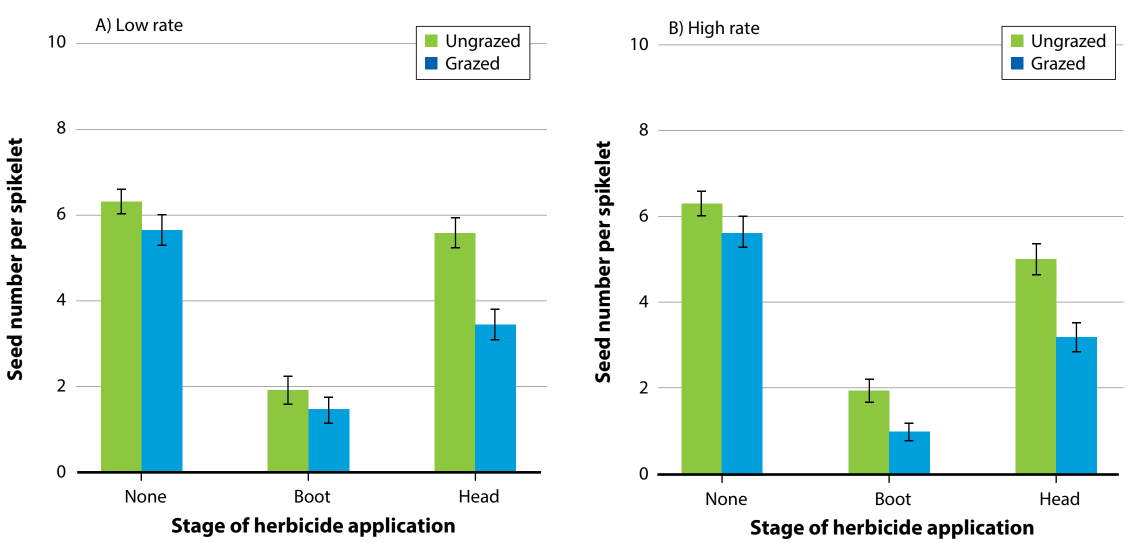


FIG. 4. Mean barb goatgrass seed number per spikelet ±SE for grazed and ungrazed treatments combined with low (A) and high (B) rates of glyphosate applied at boot and heading stages.

seed number was significantly lower compared to no grazing (fig. 4A). Seed number was also lower when high glyphosate rates combined with grazing were employed at the boot stage (fig. 4B). Seed number was lower at the boot stage (mean = 1.6 seeds per spikelet, df = 2, F = 325.4, $p < 0.001$) than it was at the heading stage (mean = 4.6). When plants were treated at tillering, barb goatgrass individuals were almost entirely absent (and seed production was effectively zero).

Interpreting results

We found that both grazing and glyphosate application are effective in reducing spikelet densities in barb goatgrass, with earlier application timings resulting in greater reductions than later application timings. Grazing was most effective when combined with later glyphosate application timings. Depending on several factors (e.g., slope and accessibility, livestock numbers available, available labor force and labor costs, fencing infrastructure, etc.), grazing treatments may be impractical on landscape scales. In such cases, intensive grazing may still be useful in targeting localized or nascent plant populations. Further, although grazing alone does not eliminate barb goatgrass or medusahead populations, reductions in spikelet density and seed number will likely translate into at least a short-term reduction in the seedbank and propagule pressure into adjacent areas (e.g., Grice 1996). This may have utility for management scenarios where chemical control options are restricted, or provide time for more intensive, integrated management efforts.

While grazing and glyphosate applications showed clear relationships in barb goatgrass, low pre-treatment density of medusahead limited our understanding of relationships among treatments and responses in this species. As with barb goatgrass, we found that earlier herbicide application timings in medusahead resulted in greater reduction of spikelet density, but we were unable to detect other differences or interactions among treatments.

Using targeted grazing in tandem with other control approaches such as low rate herbicide application can be an effective method for non-native grass control.

J Davy

Lessons for management

We found that glyphosate application at the tillering stage resulted in nearly complete elimination of spikelet density and seed production in both species, although this timing may also incur the highest long-term damage to nontarget species (Crone et al. 2009) and limit treatment utility to specific management scenarios where active revegetation or restoration is planned or fuel breaks are required. Further, glyphosate application at tillering reduced effects of grazing on spikelet density and seed production. Early glyphosate application resulted in nearly complete control and no additional benefit was observed from grazing. Although barb goatgrass plants in the two later application stages (boot and heading) were more fully developed at the time of application, the treatments still reduced spikelet densities — but only when combined with grazing. Treatment at the boot stage may provide better spikelet reduction than at the heading stage due to growth dynamics (Evans et al. 1970), but grazing and herbicide at the heading stage still result in lower spikelet densities than they do in the control. They therefore represent a viable alternative when management actions must be delayed. The additive/synergistic effect of grazing and herbicide on spikelet production and seed number integrates multiple land uses of livestock production and conservation of natural resources compromised by invasion and supports other work that has highlighted an integrative approach (e.g., Enloe et al. 2005).

Our work also suggests the possibility of using lower herbicide rates in the management of invasive grasses. We observed no differences in medusahead or barb goatgrass spikelet densities or in barb goatgrass seed number when plants were treated with different rates (high or low) of glyphosate. Using a lower rate translates to a lower treatment cost and, though not measured in this study, may also provide increased selectivity for barb goatgrass and medusahead compared to nontarget species, especially perennials (Kyser et al. 2012; Kyser et al. 2013; Morris et al. 2016). Lowering of overall herbicide usage (and glyphosate usage in particular) is an increasingly common priority for managers and jurisdictions.

While this study demonstrates the benefit of using both grazing and reduced rate glyphosate applications for barb goatgrass control, a single year of treatment should not be considered a long-term control strategy; populations could quickly rise to pretreatment levels if grazing is discontinued (James et al. 2015). At least two years of treatment are probably necessary to achieve longer-term control for barb goatgrass (Hopkinson et al. 1999), unlike for the much shorter-lived seedbank of medusahead (Blank et al. 1996; Hulbert 1955; Sharp and Hironaka 1957). For example, DiTomaso et al. (2001) has demonstrated the need for two years of controlled burning for successful control. However, residual dry matter remaining after an initial burn may not be sufficient to carry a burn in the following year

(DiTomaso et al. 2001). In such circumstances, glyphosate treatment as in the current study would be an ideal option to replace a second-year burn. An additional practical benefit of the glyphosate treatment is that barb goatgrass identification becomes much easier as the plants begin to mature, providing better targeting of specific infested areas. This translates to cost savings and lower impact on nontarget areas when compared to treating large areas.

To address the growing invasion of annual grasses across the western United States, managers should prioritize the use of integrated approaches. These approaches are known to be particularly effective for weed control on working landscapes (DiTomaso 2000), and they can reduce unintended negative effects on natives (Rinella et al. 2009). Our work demonstrates how considerations of phenology can be leveraged within an integrated pest management strategy for even more successful control of undesirable plant species on rangelands, a technique typically used in cropland systems (Knezevic et al. 2002). To help managers achieve their goals, future research should move beyond modeling exercises to deliver field-tested strategies for coupling integrated approaches with ecological considerations of phenology and life-cycle dynamics. CA

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RESEARCH ARTICLE

Growers follow the label: An analysis of bee-toxic pesticide use in almond orchards during bloom

Pesticide use data indicate that almond growers have reduced labeled bee-toxic pesticide use, but unlabeled bee-toxic agrochemicals are still applied during bloom.

by Jennie L. Durant*, Brittney K. Goodrich*, Kelly T. Chang and Evan Yoshimoto

Online: https://doi.org/10.3733/ca.2020a0030

Abstract

California almond orchards are most U.S. beekeepers' first stop on their pollination and honey production circuit, so the agrochemicals bees are exposed to in almonds can shape the vitality of their colony for the rest of the year. We explored the potential for honey bee exposure to bee-toxic agrochemicals during almond bloom by utilizing the California Department of Pesticide Regulations' Pesticide Use Report database from 1990 to 2016. We found that overall, growers are observing the pesticide labels and reducing their use of labeled bee-toxic pesticides during almond bloom. However, we also found that insect growth regulators, fungicides and organosilicone surfactants — agrochemicals often not labeled as toxic to bees — are commonly applied during almond bloom. These agrochemicals can be sublethally or synergistically toxic to adult honey bees and bee larvae, presenting potential harm to colonies during almond pollination. Our findings demonstrate the need for a shift in the U.S. Environmental Protection Agency's labeling requirements, as well as continued communication between almond growers, pesticide applicators and beekeepers to keep colonies at a low risk of bee-toxic agrochemical exposure.

Managed honey bees add an estimated $17 billion in direct and indirect pollination services to nearly 70% of all major food crops in the United States (Calderone 2012). Yet despite their critical economic and ecological role, the current state of honey bees is precarious. In 2018, commercial beekeepers (those managing 501 or more colonies) lost over 37.5% of their colonies during winter, while stating that losses of less than 22% were economically viable (Bruckner et al. 2019). Research indicates that honey bee vulnerability is due to a nexus of stressors: parasites such as *Varroa destructor* mites and the gut fungus *Nosema ceranea*, pathogens and disease, a lack of healthy and diverse pollen resources, and exposure to bee-toxic pesticides (Goulson et al. 2015). In 2016,

An analysis of pesticide use data from 1990 to 2016 shows that San Joaquin Valley almond growers have reduced their use of labeled bee-toxic pesticides during almond bloom. However, documented bee-toxic agrochemicals without EPA precautionary statements are still commonly applied during bloom.

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commercial beekeepers in the United States attributed approximately 9% of their colony losses to pesticides (Kulhanek et al. 2017).

Each year, the majority of U.S. beekeepers truck their honey bees to California to pollinate almonds from mid-February to mid-March. California growers produce 100% of the almonds commercially grown in the United States (CDFA 2017, 113), and over 80% of global almond production (Almond Board 2017, 7). This demand has contributed to almonds' high market value and subsequent expansion from over 480,000 planted acres in 1995 to 1.3 million acres in 2018 (CDFA 2018). As the almond industry has expanded, it has also required an increasing number of honey bee colonies. The current recommendation is two colonies per acre for maximum pollination services (USDA and FCIC 2018). This meant that, in 2018, approximately 2 million colonies were required in California's Central Valley to pollinate almonds — around 81% of managed honey bee colonies in the United States (Goodrich et al. 2019).

Pollinating for the almond industry has benefits and challenges for beekeepers and their honey bees. Almond pollination fees have significantly increased over the years, and the income from almond pollination now provides over a third (33.7%) of all beekeeper revenue in the United States (Ferrier et al. 2018, 6). Almond pollen is also high in protein (Ellis et al. 2013) and good for the development of young honey bee workers (Keller et al. 2005a and 2005b). However, the honey that bees produce while in almonds is bitter and largely unmarketable, and preparing bees for an early February pollination is labor and input intensive for beekeepers (Durant 2019). Managed colonies may also be exposed to bee-toxic agrochemicals during almond bloom that can have toxic effects on honey bees (Fisher II et al. 2017, 2018; Wade et al. 2019).

Almonds are most beekeepers' first stop on their annual pollination and honey production circuit, so the agrochemicals bees are exposed to in almond orchards can shape the vitality of their colony for the rest of the year, affecting their ability to meet future pollination contracts and earn income from honey production. To reduce pesticide exposure to honey bees, growers are encouraged by UC Integrated Pest Management (UC IPM), the California Department of Pesticide Regulation (CDPR) and the Almond Board of California (Almond Board) to primarily focus their use of insecticides during the dormant period before bloom begins in mid-February (Almond Board 2014; CDPR 2018; Pickel et al. 2004). At the same time, however, growers sometimes find it necessary to use fungicides, insect growth regulators or other pesticides during bloom — agrochemicals that can be sublethally or synergistically toxic for bees but are not labeled as such (see online technical appendix, table A).

Our objective was to trace growers' application of these chemicals during almond bloom and assess the potential risks of pesticide exposure for honey bee colonies. We evaluated growers' pesticide use in the San Joaquin Valley, the region where the greatest number of almond orchards (CDFA 2019), and subsequently bee colonies, concentrate each year (Goodrich 2017).

Federal pesticide regulation

In the United States, pesticide regulation is overseen by the U.S. Environmental Protection Agency (EPA), which reviews the product label as part of the licensing and registration process for pesticides as mandated by the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) and the Code of Federal Regulations (40 CFR, parts 150–189) (U.S. EPA 2019a). Every pesticide product is required to have a hazard and precautionary statement for environmental hazards, including risks to non-target insects (Labeling Requirements for Pesticides 2001, subpart E). The hazard statement describes the type of hazard that might be present, while the precautionary statement instructs which actions the user must take to "avoid the hazard or mitigate its effects" (Labeling Requirements for Pesticides 2001, subpart E).

To register a pesticide for outdoor use, FIFRA mandates that companies must provide EPA with reliable data on its toxicity for honey bees, including the results of an adult honey bee acute contact test (Data Requirements for Pesticides 2007, subpart G). The acute contact test is designed to determine the median lethal dose of a pesticide — either the end product formulation or active ingredient — that will kill 50% of an experimental population of adult honey bees through a topical application of the test substance (i.e., an LD_{50} value) (US EPA and OCSPP 2012). After bees'

In 2018, approximately 2 million honey bee colonies were required in California's Central Valley to pollinate almonds — around 81% of managed honey bee colonies in the United States.

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The EPA's moderate and highly toxic ratings require a "Bee Hazard" graphic on the product label.

thoraxes are exposed to the substance, bee mortality is observed for a maximum of 96 hours. If the LD_{50} dose is 11 micrograms (μg) or more, or bees will never encounter the substance (e.g., rat poison), then the pesticide is considered essentially non-toxic to bees. If the value is between 2 μg and 10.9 μg, the pesticide is labeled "moderately toxic", and if the LD_{50} dose is less than 2 μg, the pesticide is considered highly toxic (US EPA 2016). Both the moderate and highly toxic ratings require a "Bee Hazard" graphic on the label.

Highly and moderately toxic pesticides can have a range of effects on bees but are not the only source of high-impact exposure. These pesticides often kill bees on contact (Medrzycki et al. 2013), but poisoned bees can also become irritable and likely to sting, tremble, become paralyzed, or exhibit other abnormal behavior. In general, the symptoms occur quickly (within 96 hours) and are easy to observe. The symptoms of sublethal pesticide poisoning, which are not considered "toxic" per EPA standards, are often more complex or may take longer to become apparent. These symptoms might include decreased learning ability and foraging, decreased brood production and egg laying by the queen, or emergent bee wing deformation and stunted growth (Thompson 2003). In general, a worker bee must be able to fly, use short and long-term memory functions to communicate, care for larvae, and perform other social functions (Medrzycki et al. 2013). Determining sublethal or chronic toxicity requires different assays or test procedures to gauge the effect of the test substance on these functions, a more complex process than acute toxicity tests.

Though FIFRA only mandates the acute contact test, EPA typically requires more extensive testing as part of registration (Douglass and Steeger 2019; Housenger and Douglass 2019). Around 2016, EPA began to require the oral toxicity test, where adult bees are fed the test substance for acute toxicity, as well as a 21-day honey bee larval toxicity test (OECD 2016; US EPA 2016). EPA may also require some chronic toxicity testing in the future, depending on the active ingredient and its end use.

EPA has begun to include larval toxicity on labels when warranted, though labels do not currently include information on sublethal toxicity (US EPA 2012). As a result, agrochemicals that are sublethally toxic to adult or larval bees may be registered without a bee-toxic precautionary label. In addition, EPA only began to require the acute oral toxicity test and the 21-day larval toxicity test in 2016 (US EPA 2016). Chemicals registered before this requirement in 2016 may not have required these data and their labels may not reflect acute oral or larval toxicities.

Pesticide regulation in California

In addition to the rules imposed by EPA, California has its own authority to regulate and license pesticides, guided by FIFRA, section 24(a), and a number of California laws and regulations (CalEPA and CDPR 2015, 4). The California Environmental Protection Agency (CalEPA) and CDPR oversee pesticide enforcement in collaboration with each county's agricultural commissioner's office (CAC). If growers plan to apply a labeled "bee-toxic" or "restricted use" chemical, they are required to contact the CAC 48 hours prior to application and then notify beekeepers to give them time to move their bees, cover them or discuss an application plan that will best protect their bees. If the pesticide does not have a bee-toxic precautionary label, the grower is not required to contact the beekeeper.

This is a challenge for beekeepers, because some of the pesticides used during dormancy and bloom, such as fungicides and insect growth regulators (IGRs), have demonstrated toxicity to bees (see technical appendix, table A). Many of these chemicals tend to affect honey bee larval development, and as a result, beekeepers often do not notice damage from these agrochemicals until weeks after the application. Shortly after almond bloom ends around mid-March, beekeepers may relocate their hives for honey production, spring splitting (the process of expanding colony numbers) or for their next pollination contract. This can make it difficult to pinpoint which chemical caused the damage, and especially challenging to report it.

Unlabeled agrochemical toxicity

An additional challenge beekeepers face is that some pesticides exhibit synergistic toxicity when mixed with other agrochemicals in the sprayer tank (Fine et al. 2016; Mullin 2016; Wade et al. 2019; Zhu et al. 2014; see technical appendix, table A, for more citations). Pesticide synergy occurs when the combination of two or more active ingredients are more powerful (e.g., more toxic to bees) than the effects the chemicals would have individually (US EPA 2019b). Growers or pesticide applicators often tank mix to reduce application costs by limiting the number of times a spray rig must go through an orchard, so they might combine all the desired chemicals in one tank and spray them on one application trip. Controlled studies indicate that some insecticides and fungicides used in almond orchards become more toxic to bees when mixed than if the chemicals were applied separately (Fisher II et al. 2017; Wade et al. 2019). Currently, synergistic interactions are not addressed on pesticide labels. EPA is finalizing a process in which registrants are required to document whether they have any existing patent claims for synergistic activity (US EPA 2019b), but the implications of this new process for labels is unclear.

Pesticides are not the only problematic agrochemical for bees. A class of agrochemicals called organosilicone surfactants, a type of spray adjuvant, can make bee larvae more susceptible to viral pathogens and decrease olfactory learning in honey bees (Fine et al. 2016), which may have implications for honey bee foraging abilities (Ciarlo et al. 2012). Adjuvants are spray tank additives that enhance the ability of pesticide formulations to help them spread or stick to the foilage of the target plant or the surface of the target insect (US EPA and OCSPP 2015). Formulations with organosilicone surfactants are more likely to penetrate honey bees' waxy cuticle and — perhaps most importantly — can increase the toxicity of other chemicals (May et al. 2015).

Because adjuvant products "don't make pesticidal claims," they are not considered pesticides by EPA, and thus will not be labeled with a precautionary statement or tested for their bee toxicity (US EPA and OCSPP 2015). CalEPA and CDPR, however, define a spray adjuvant as a pesticide, require its registration (CalEPA and CDPR 2015, 5), and require that any applications be reported to the county agricultural commissioners' office. However, like EPA, CDPR also does not require ecotoxicology testing, including for pollinators, on adjuvant products, which means that their actual toxicity is unknown to regulatory agencies. This puts many growers, and the pesticide applicators who apply pesticides in their orchards, in a position where the labels and regulations they rely on to safely apply pesticides do not reflect their actual toxicity to bees.

Methods

To investigate grower pesticide use, we drew from CDPR's Pesticide Use Report (PUR) database for each of the eight counties in San Joaquin Valley (Fresno, Kern, Kings, Madera, Merced, San Joaquin, Stanislaus and Tulare) from 1990 to 2016 (CDPR n.d.). We statistically analyzed trends throughout the whole time period, but given the technological changes that generate new classes and formulations of pesticides, we largely focused figures and discussion on pesticide use from 2000 to 2016. To account for expanding almond acreage, we used each county's annual crop report data to gather bearing acreage of almonds by county. We divided pounds of active ingredients and applications by bearing almond acreage to adjust for increased applications over the time period due to the increased acreage.

Our analysis concentrated on agrochemical applications in almonds during almond bloom each year (February 15 to March 15), and during the months when bees would commonly be in California either before (January 1 to February 15) or after bloom (March 15 to April 1). By January 1, 2018, roughly 761,000 colonies had already been shipped into California for almond bloom, and during the month of January 2018 another 633,000 colonies were shipped in (CDFA, unpublished data).

To obtain the pesticide toxicity and label status, we referred to the UC IPM website on "Bee Precaution Pesticide Ratings" (UC IPM n.d.) and the Pacific Northwest Extension publication "How to Reduce Bee Poisoning from Pesticides" (Hooven et al. 2016). The latter compiles commonly used pesticides and organizes them by toxicity; it also indicates pesticides which have no precautionary statements but require further data. We then conducted an extensive literature review of pesticides used during bloom that do not have precautionary labels. Table A in the technical appendix notes the effects of many of these non-acutely toxic agrochemicals on honey bees, though it is not exhaustive. We define the category of non-acutely toxic chemicals to include chemicals that peer-reviewed research indicates are sublethally or synergistically toxic to bees, but do not have an EPA acute toxicity rating. Tables B and C (technical appendix) show the agrochemicals analyzed in this paper.

Additionally, this paper was informed by over 81 semi-formal interviews with almond-pollinating beekeepers with operations ranging from 20 colonies to over 20,000 colonies, almond growers, researchers and extension specialists working with beekeepers and almond growers and government officials from the county agricultural commissioner's office and EPA (Durant 2019). Unless specified as "commercial beekeeper," our use of the term "beekeeper" refers to any beekeeper that pollinates almonds, of any operation size.

We analyzed historical trends using linear regression analysis. Full regression results are reported in technical appendix tables D, E and F. The regressions were performed on applications and active ingredients applied during bloom, summarized at the county level each year. The regressions measure the average trend of the dependent variable (either applications or active ingredients per acre) over the 1990–2016 time period. We supplemented the statistical trend analyses with figures and discussions that represent more recent time periods.

Results of pesticide analysis

We found that since 1990, both the bloom-time (February 15 to March 15) applications per acre and the amount per acre (pounds of active ingredient) of the pesticides listed in table B have decreased (figs. 1 and 2). Using regression analysis, both of these trends were statistically significant at the 1% level (technical appendix tables D and E). Since 1990, the amount of


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active ingredient applied per bearing acre of almonds has decreased on average across counties by 0.09 pounds per year (fig. 1). Applications per bearing acre of almonds per year have decreased on average by 0.0001 applications per year (fig. 2). The decreasing trend in applications per acre is less apparent over the 1990–2016 time period, though applications per acre since 2010 have decreased substantially. The decreasing trends in applications and active ingredient per acre vary across type and toxicity of pesticides and are discussed below.

Types of agrochemicals applied

Fungicides were the only pesticide category with a significant decrease in pounds of active ingredient applied during almond bloom over the 1990–2016 time period (p-value < 0.01; fig. 3). Since fungicides make up the majority of figure 3, with herbicides being the

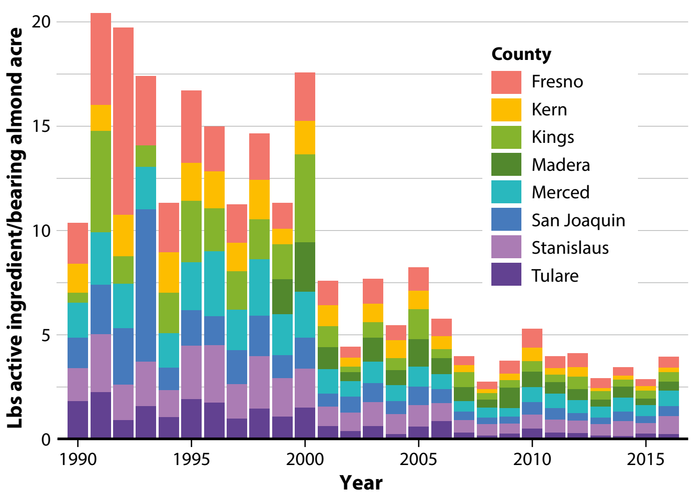

FIG. 1. Pounds of active ingredient of agrochemicals in table B applied per bearing almond acre during almond bloom by county, February 15 to March 15, 1990–2016.

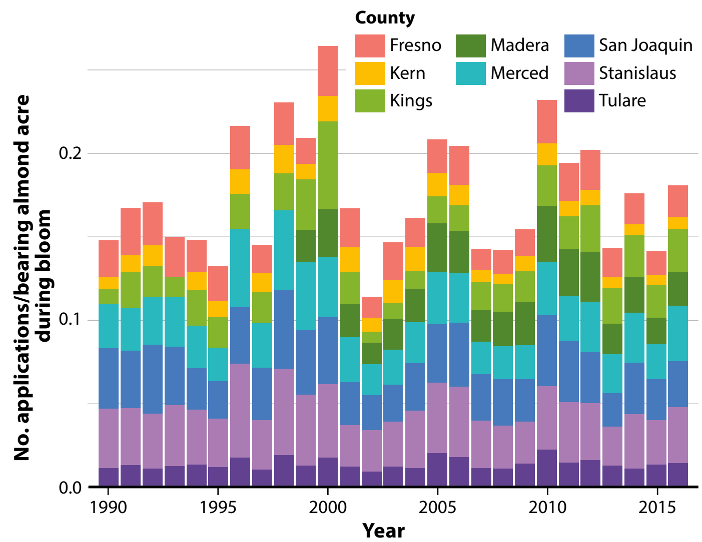

FIG. 2. Applications of all agrochemicals in table B per bearing almond acre during almond bloom by county, February 15 to March 15, 1990–2016.

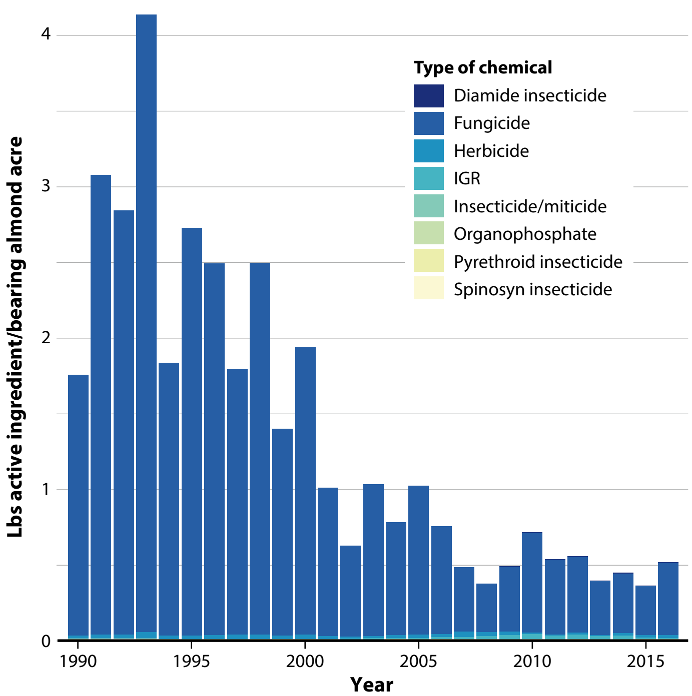

FIG. 3. Pounds of active ingredient of agrochemicals in table B applied per bearing acre of almonds during bloom by type of chemical, February 15 to March 15, 1990–2016. This figure largely only represents fungicides and to a lesser extent, herbicides. Note: Listed miticides are those applied to almond orchards, not to treat *Varroa* mites on honey bees.

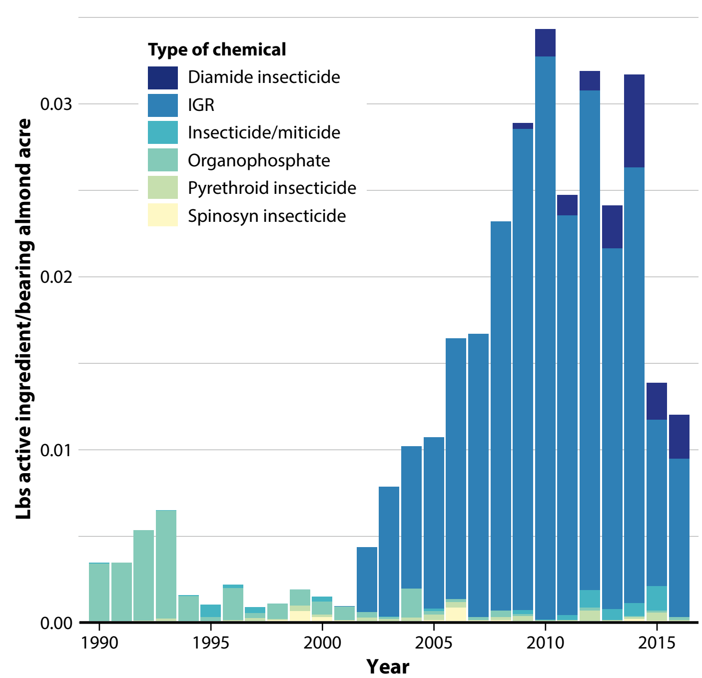

FIG. 4. Pounds of active ingredient of agrochemicals in table B applied per bearing acre of almonds during bloom by type of chemical, without fungicides and herbicides, February 15 to March 15, 1990–2016. Note: Listed miticides are those applied to almond orchards, not to treat *Varroa* mites on honey bees.

next largest category, Figure 4 shows the pounds of active ingredient applied by type of chemical, excluding fungicides and herbicides. From figure 4, it is apparent that there was a switch in the late 1990s from the use of highly toxic insecticides, organophosphates and pyrethroids, to IGRs. The use of IGRs increased from 2002 to 2010, when organophosphate and pyrethroid usage began to decrease.

Time periods, toxicity of applications

Highly toxic chemicals did not have statistically significant trends in applications or active ingredient applied per acre over the 1990–2016 time period (technical appendix tables D and E). Since 1990, pounds per acre of active ingredients applied have decreased on average by 0.087 ($p < 0.01$) and 0.007 ($p < 0.10$) for non-acutely toxic and moderately toxic chemicals, respectively. There was no statistically significant trend in applications per acre of non-acutely toxic chemicals since 1990, while applications per acre of moderately toxic chemicals have decreased ($p < 0.01$). Consistent with the absence of a long-term trend, highly toxic chemicals have been applied at low levels per acre during almond bloom since the year 2000, so there was little room to decrease this amount over time (fig. 5). Moderately toxic chemicals were applied at slightly higher levels per acre than highly toxic ones and saw a slight decrease between 2000 and 2016. Non-acutely toxic chemicals were applied at relatively high levels per acre beginning in 2000, but have decreased over time (fig. 5).

Table 1 shows the average number of agrochemical applications per day in the San Joaquin Valley for

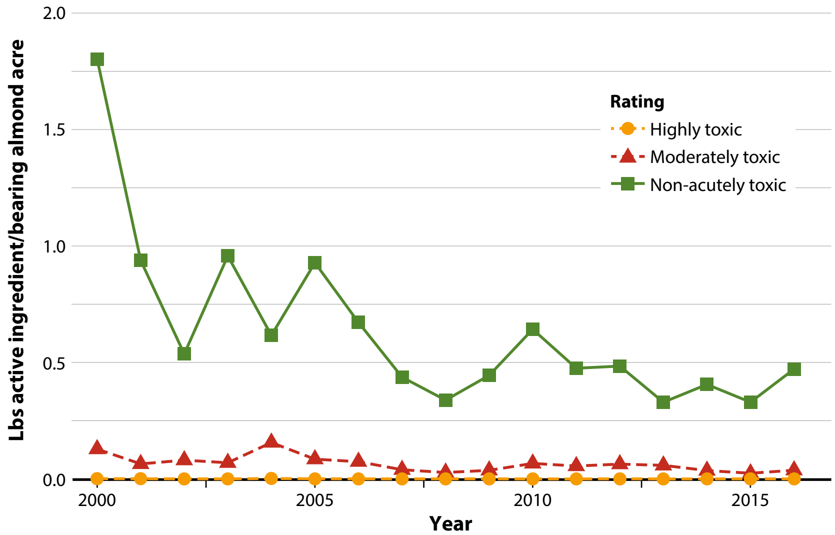

FIG. 5. Pounds of active ingredient of agrochemicals in table B applied per bearing almond acre during almond bloom by bee-toxicity rating, February 15 to March 15, 2000–2016. Note: Non-acutely toxic includes sublethally and synergistically bee-toxic chemicals (table A).

TABLE 1. Mean number of total applications per day in San Joaquin Valley for agrochemicals in table B by toxicity rating and timing, 2010–2016

	Mean no. of total applications per day		
Bee toxicity rating	**Pre-bloom (Jan 1–Feb 14)**	**Bloom (Feb 15–Mar 15)**	**Post-bloom (Mar 16–Apr1)**
Highly toxic	49	1	10
Moderately toxic	31	21	3
Non-acutely toxic	51	594	276

Note: Non-acutely toxic includes sublethally and synergistically bee-toxic chemicals (online appendix table A).

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the years 2010 to 2016, separated by bee-toxicity rating (highly toxic, moderately toxic, and non-acutely toxic) and timing with respect to almond bloom. This table is not adjusted for almond acreage, and is meant to broadly reflect the growers' decisions regarding pre-, during- and post-bloom applications. The most harmful chemicals are applied infrequently when bees are in almond orchards during bloom, on average one application per day. However, there are still some applications of highly toxic and moderately toxic pesticides prior to bloom. Non-acutely toxic chemicals (primarily fungicides, IGRs and herbicides) are often applied within the almond bloom time period, and it is clear that non-acutely toxic applications compose the majority of all chemical applications during bloom with an average of 594 applications per day.

Figures 6 and 7 show histograms of chemical applications with and without a bee-toxic precautionary statement, respectively. We found that chemicals with precautionary statements were applied before bloom, but rarely during the bloom period (fig. 6). On the other hand, chemicals without precautionary statements were frequently applied during bloom (fig. 7). These results highlight one of our key findings, that growers are following the label during bloom, but are also applying agrochemicals without precautionary statements (which are sometimes bee-toxic) while bees are pollinating almonds.

Our results indicate there is no noticeable trend with applications per acre of pesticides with no precautionary statement; however, the trend in pounds per acre of active ingredient since 1990 is negative and statistically significant ($p < 0.01$; fig. 8). The average number of applications per acre with chemicals labeled with precautionary statements has decreased over time ($p < 0.01$), though there was no statistically significant trend in active ingredient per acre. Declines in applications per acre during bloom may be due to a greater awareness among almond growers and pesticide applicators

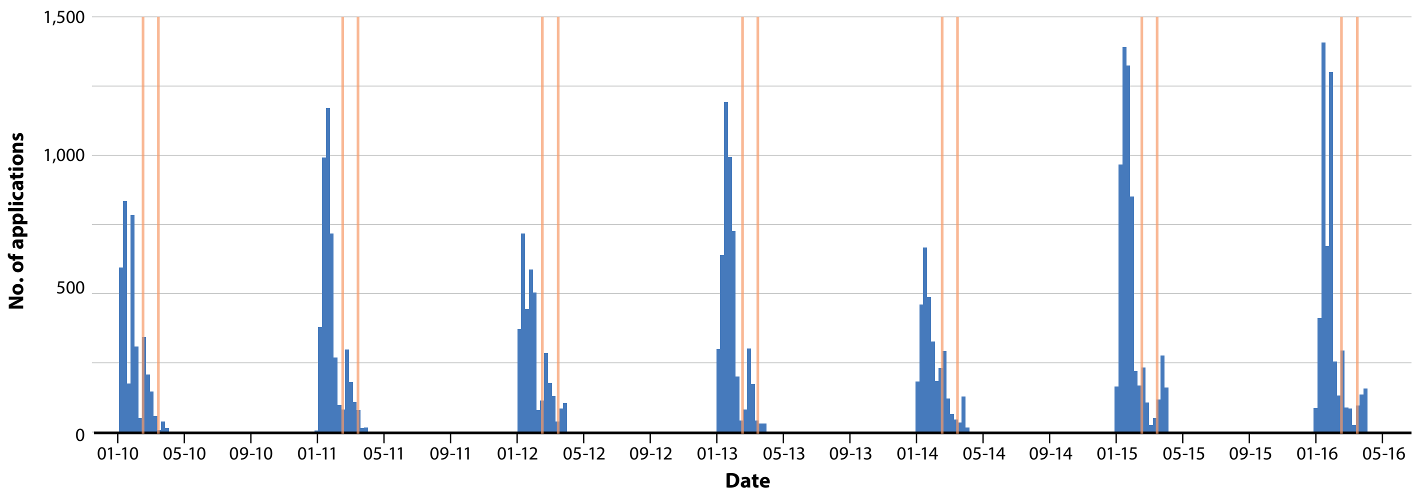

FIG. 6. Histogram of weekly applications of agrochemicals in table B with precautionary statements (almond bloom period highlighted), January 1 to April 1, 2010–2016.

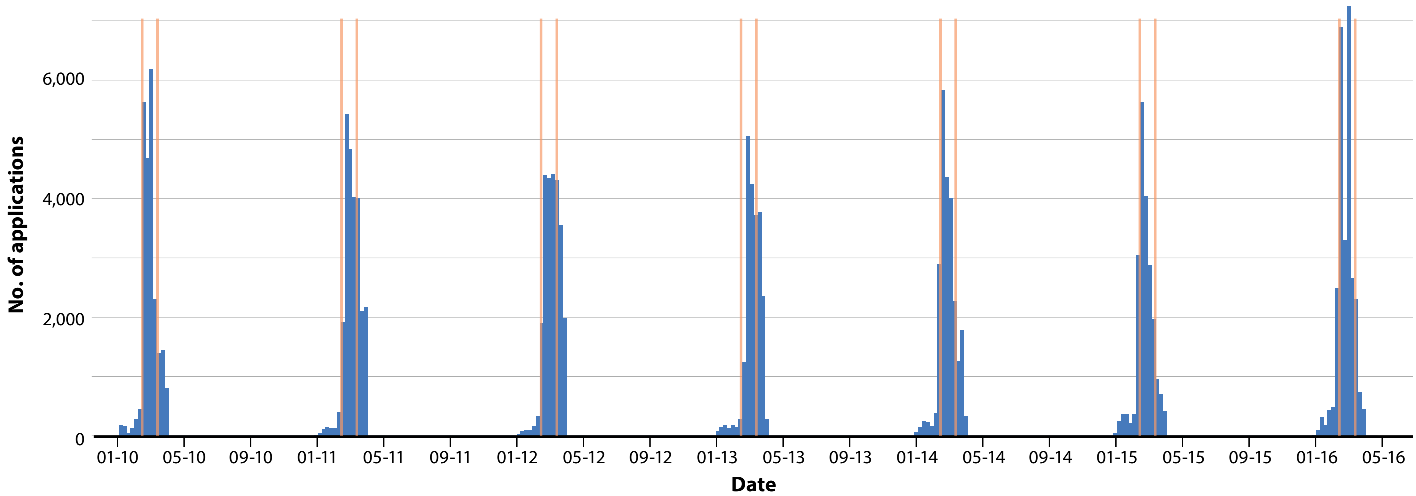

FIG. 7. Histogram of weekly applications of agrochemicals in table B without precautionary statements (almond bloom period highlighted), January 1 to April 1, 2010–2016.

that these pesticides can be toxic to bees, or it may just indicate sensitivity to label instructions.

Organosilicone surfactant applications

Since 1990, there has been an upward trend in organosilicone surfactant use in pounds per acre during almond bloom ($p < 0.01$); this seems to vary by toxicity rating (technical appendix table F). Over the 1990–2016 period, the increasing trend for sublethal organosilicone surfactants was more than that of surfactants with unknown toxicity. Figures 9 and 10 suggest that organosilicone surfactant use per bearing acre of almonds during bloom has increased in recent years, especially in application levels of organosilicone surfactants that have unknown toxicity to bees. The majority of organosilicone surfactants applied may have sublethal toxicity to bees, though more research needs to be conducted to understand the depth and breadth of their toxicity (Chen et al. 2018; Mullin et al. 2016). It would be interesting to know why the use of organosilicone surfactants was so low in 2012. This information might provide useful insights into tactics that would decrease the use of these chemicals.

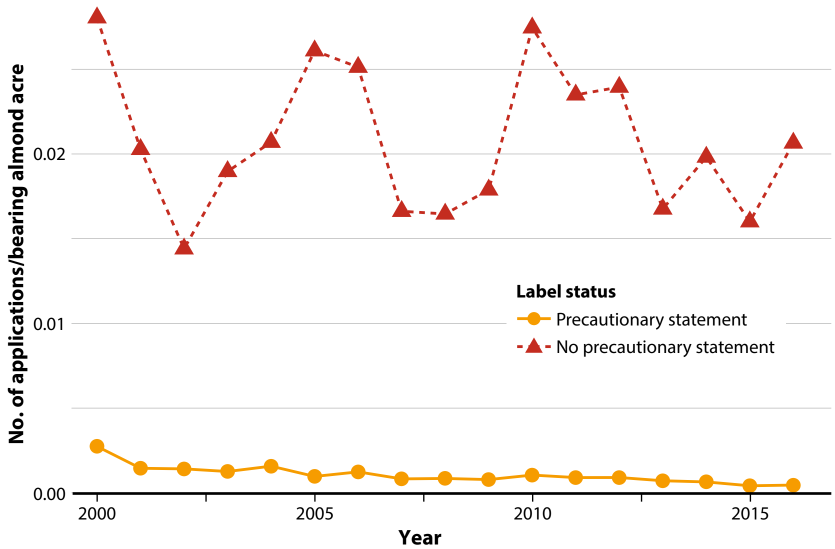

FIG. 8. Applications per bearing almond acre of agrochemicals in table B during almond bloom by label status, February 15 to March 15, 2000–2016.

Study limitations

One limitation of our analysis is that we did not analyze the time of day agrochemicals were applied. The time of application can make a big difference in bee toxicity due to honey bees' typical foraging behavior. This is a knowledge gap that limits our ability to interpret our findings. For example, bee-toxic agrochemical use could be rising, but if growers are applying these chemicals only in the late afternoon or evening then it may not be problematic for bees, who typically stop foraging by that time. Timings of agrochemical applications are included in the PUR data and are an area for future exploration.

A second limitation is that this study only looks at agrochemical applications by almond growers, while many beekeepers express concern about pesticide damage from neighboring crops such as stone fruits or alfalfa. This study does not address agrochemical use from neighboring crops in the region, which is another area for future research. A third limitation is that we do not know definitively if the decreasing trends in agrochemical use is due to the increased toxicity of the formulation, which then requires less pounds per acre for each application. We suspect this is the case, but is outside the scope of this study and another area for future research.

Steps for mitigating bee losses

To address the use of sublethally and synergistically bee-toxic pesticides during almond bloom due to the knowledge gap in the EPA labeling system, beekeepers, extension specialists, the Almond Board, EPA and CDPR jointly crafted and publicized a set of Honey Bee Best Management Practices (Bee BMPs) in 2014 (Almond Board 2014). The Bee BMPs have four core

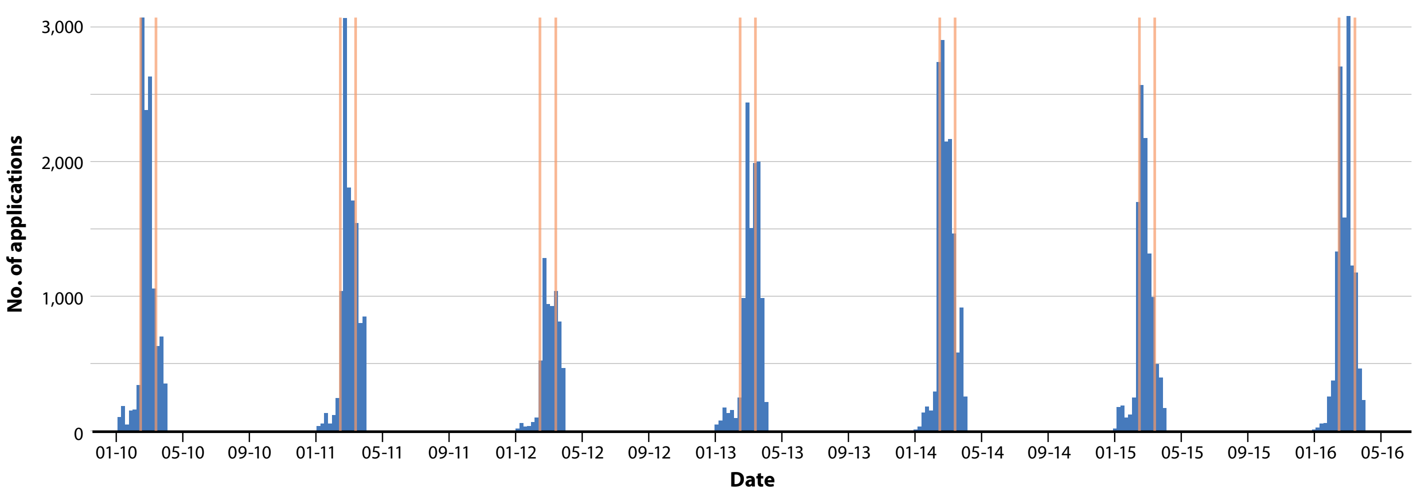

FIG. 9. Histogram of weekly applications of organosilicone surfactants in table C (almond bloom period highlighted), January 1 to April 1, 2010–2016.

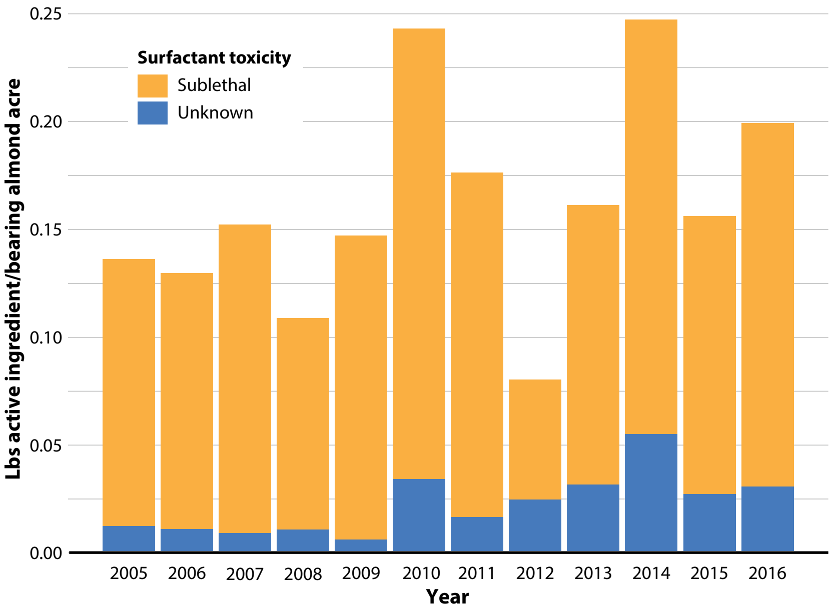

FIG. 10. Pounds of organosilicone surfactant (table C) applied per bearing acre of almonds during bloom by toxicity rating, February 15-March 15, 2005–2016.

precautions: (1) maintain communication between all parties on the specifics of pesticide application, specifically beekeepers and/or bee brokers, pesticide control applicators, farm managers and land owners; (2) only spray fungicides in the late afternoon or evening; (3) avoid tank-mixing products during bloom because some agrochemicals might have synergistic toxicities for honey bees and (4) avoid applying all insecticides during bloom. The dataset in this paper only reflects 2 years (2015–2016) of grower pesticide practices after the introduction of the Bee BMPs in 2014, so we do not attempt to make conclusions about the efficacy of its dissemination and the adoption rate of the BMPs. This would be an interesting topic for future research.

In addition to our finding on bloom-time applications of chemicals, we found that many bee-toxic pesticides are applied in January, the month before almond bloom (table 1). This is problematic for beekeepers, given that many beekeepers store their colonies in or near almond orchards over winter and the remaining colonies usually arrive by the beginning to middle of January. The applications of highly toxic and moderately toxic pesticides prior to bloom (fig. 6) may still have an impact on these colonies, and may deter beekeepers from bringing colonies into California early. This may have some unintended consequences for almond growers: in years when almonds bloom earlier than normal, and/or if a large number of beekeepers delay entry into California creating bottlenecks at California border protection stations, growers may not have all the colonies needed for adequate pollination of early blooming varieties.

Other efforts aim to facilitate better communication around pesticides between beekeepers, growers and the counties. In 2014, the Almond Board began disseminating UC Cooperative Extension research demonstrating the efficacy of insecticide applications outside of almond bloom (Almond Board 2014, 4). In 2017, a coalition of stakeholders led by the California Association of Pesticide Advisors and the County Agricultural Commissioners and Sealers Association partnered to create the BeeWhere program (https://beewhere.calagpermits.org/). Launched in 2019, BeeWhere offers an online portal with a GIS mapping system where beekeepers can register their hives with the county, so growers and pesticide applicators can notify them before an agrochemical spray. Beekeepers and almond growers may also consider mitigating issues caused by pesticide damage through the use of clauses in their almond pollination contracts. In fact, in a survey of growers attending the 2015 Almond Conference, roughly 30% of respondents included pesticide clauses in their pollination contracts (Goodrich 2017).

Factors impacting honey bee colony health during almond pollination can have a major influence on bee health throughout the United States for the remainder of the year, and are important to trace given honey bees' crucial role in our agricultural system. Our findings suggest that changing EPA labeling requirements to include sublethally and synergistically bee-toxic agrochemicals, registering adjuvants as pesticides with the EPA and requiring larval and chronic toxicity tests as part of this registration, and growers' full adoption of the Almond Board's Bee BMPs, may all be important steps toward improving bee health. CA

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This research was supported by the USDA National Institute of Food and Agriculture NIFA fellowship, project #1019243.

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Online UC ANR webinars and classes

UC Ag Experts Talk: Biology and Management of Avocado Lace Bug in California

https://ucanr.zoom.us/webinar/register/WN_cLBzzsbZS7GIgiDtSxqWBA

Date: July 7, 2021
Time: 3:00 p.m. to 4:30 p.m.
Location: Online (Zoom)
Contact: Petr Kosina, UC IPM pkosina@ucanr.edu

Food Growing Forum July Topic: Pests of the Summer Garden

http://ucanr.edu/2021FoodForumJuly

Date: July 11, 2021
Time: 3:00 p.m. to 4:00 p.m.
Location: Online (Zoom)
Contact: UC Master Gardeners of Napa County, mastergardeners@countyofnapa.org

NorCal Climate Futures: Water Resources in Northern California

https://ucanr.edu/survey/survey.cfm?surveynumber=33209

Date: July 15, 2021
Time: 6:00 p.m. to 7:30 p.m.
Location: Online (Zoom)
Contact: Hannah Bird, Hopland Research and Extension Center hbird@ucanr.edu