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Editors for this issue

JOEL TRUMBO was a long-time editor for the Journal. He started his career with CDFW in 1991 as an environmental scientist working for the Pesticide Investigations Unit (PIU). His primary responsibility was to oversee the use of herbicides on department-managed lands statewide. This role required a fairly specialized understanding of invasive weed biology, herbicides and non-chemical weed control strategies, and a solid understanding of pesticide impacts on wildlife. Apparently, Joel liked the work, because he remained in that role for nearly his entire career with the department. Joel's position was eventually transferred to the Lands Program of the Wildlife Branch. In 2018, Joel took on the Environmental Program Manager role for the Lands Program, a position he held until his retirement in February 2020.

JENNIFER OLSON was a guest editor for this issue. She is a Senior Environmental Scientist with CDFW's Coastal Habitat Conservation Planning group in the Eureka field office. She has worked for the Department since 2013 in a variety of roles, primarily focused on environmental review and permitting. She currently serves as the Caltrans Liaison for Del Norte, Humboldt, and Mendocino Counties. Prior to working for the Department, she worked as a Research Associate for the Montana Cooperative Wildlife Research Unit where she supervised field crews and data management for projects focused on life history variation in songbirds in the U.S, Venezuela, and Malaysian Borneo. Jen is originally from Minnesota and has a bachelor's degree in Environmental Studies from the University of Minnesota-Duluth. In her free time, she enjoys birding, running, finding new places to go hiking and backpacking with her husband and her dog, and expanding her natural history knowledge about her Northern California home.

ROBERT SULLIVAN is still a relatively new member of the Journal's editorial team—he joined us in spring of 2019, but he jumped in with both feet. In just over a year, Bob has served as Associate Editor (AE) for four manuscripts (about double the normal amount for an AE)—and that's in addition to submitting four manuscripts to the Journal! He completed his first three degrees at Humboldt State University: a B.S. in Biology, an M.S. in Biology, and an M.S. in Wildlife and Natural Resource Management. He then did his PhD in Biology at the University of New Mexico and post-doctoral research as the Curator of Mammals at the New Mexico Museum of Natural History. Dr. Sullivan has had a diverse research background in everything from salmonids to herpetofauna to rodents and marine mammals, among others. He began his career with CDFW in 2007 as an Environmental Scientist in the Timberland Planning Program, and in 2009 moved to the Wildlife Program at the North Coast Wildlife Area Complex, which he continues today.

MARIO KLIP is the newest member of the Journal's editorial team. He joined the Department in 2012 with the North Central Region. He obtained a PhD in Environmental Sciences Policy at the University of California, Berkeley in 2018, with a focus on black bears, and he completed his master's in Biology at Sonoma State University in 2012. Dr. Klip had a very different career before joining CDFW; prior to working with wildlife and conserved lands, he worked in the IT industry, and he holds a bachelor's in International Management and Marketing and master's in Management Accounting from the University of Amsterdam. He moved from the Netherlands to California for various projects and ended up in Silicon Valley. Dr. Klip held several senior positions before drastically changing careers to pursue a life-long desire to focus on wildlife. He is passionate about conducting applied research to better inform wildlife and land use decisions.

California Fish and Wildlife, Cannabis Special Issue; 8-11; 2020

Introduction

RYAN MATHIS, *Environmental Program Manager, Cannabis Permitting Program, Habitat Conservation Planning Branch, California Department of Fish and Wildlife*

When I moved to Fieldbrook, California in the winter of 1995 to study wildlife management at Humboldt State University, I inadvertently rented a room from someone that “grew” with friends on a property in southern Humboldt County (one of the three counties collectively known as “the Emerald Triangle”). Keep in mind that the Compassionate Use Act that would ultimately lead to what we know as Proposition 215 would not be on the ballot until November of 1996. Cannabis was not the sole source of income for the landowners, and their gardens were small, temporary, easily moved, and more importantly out of sight of helicopters. It is probably fair to assume that federal and state prohibitions guided those cultivation practices. Therefore, my first impression of cannabis cultivation was a secretive and low-impact practice carried out by conservation-minded individuals.

This secluded, low-impact cultivation trend had been the norm among cannabis growers in Humboldt County since it became the nation’s most prominent center for production of the crop in the mid-1970s. This was a result of the “Back-to-the-Land” movement that began in the 1960s with many young people leaving urban cities like San Francisco to live in more rural areas. About the same time that growing cannabis was becoming popular in Humboldt and the surrounding counties of Mendocino and Trinity (the Emerald Triangle), the U.S. government unintentionally aided in creating a market for the crop by supporting Mexico’s government in using a toxic herbicide on Mexican cannabis[1]—up until this time, most of the cannabis used in the U.S. came from across the border. This combined with the increased demand for cocaine trafficking[2]—a much more profitable export compared to cannabis—from Mexico in the late-1970s led to the Emerald Triangle becoming the mecca for cannabis production. The popularity of cocaine use in the U.S. in the 1980s most likely created a niche for the Emerald Triangle to fill the void of the mass produced cannabis from Mexico (e.g., seeds adapted to growing in the tropics) with a higher quality product (e.g., seeds adapted to growing at similar latitudes) from the Emerald Triangle.

Fast forward to 2002-2007, and gardening supply shops were popping up around Eureka (the largest city in Humboldt County), and mega grows started showing up on the landscape. Simultaneously, financing was readily available to anyone, regardless of their financial stability, who wanted to buy real estate. Those lending practices eventually led us to an economic crisis in 2008. Banks were failing, businesses were closing, and the federal government was forced to bail out lenders and auto manufacturers. While the economy was collapsing around, some growers flourished as it appeared nothing was being regulated (e.g., land grading, timber harvest and forest clearing, water diversion, water storage) in areas of Humboldt County that some would say were finally starting to recover from the logging practices of the 1950s and 60s. By 2010, it was estimated that nearly 80% of the nation’s cannabis came from California (and most of that from the Emerald Triangle). Cannabis is now a multibillion-dollar industry in California with at least 50,000 farms in the state (both illegal and legal).

The California Department of Fish and Wildlife (CDFW) took the lead on assessing environmental damage from unregulated cannabis cultivation, and began laying the foundation for a team of scientists and wildlife officers that would be dedicated to reducing impacts to fish and wildlife resources resulting from cannabis cultivation. CDFW scientific and enforcement staff began noticing large cannabis farms in the forests of the Emerald Triangle, with bulldozed redwoods forests cleared to make room for the crop. This eventually led to the creation of the Watershed Enforcement Program—a team of scientists, law enforcement officers, and attorneys charged with reducing the environmental damaged caused by cannabis cultivation on public and private lands in California. Between 2013 and early 2018, more than 700 inspections resulted in 399 tons of trash removed from public and private lands including: 2.4 million feet of irrigation pipe, 50 tons of fertilizer, and 465 gallons of chemicals (many illegal in the U.S.). In addition, the removal of 709 illegal dams and water diversions resulted in restoration of 800 million gallons of water back into local watersheds.

In 2015, the Governor approved the Medical Cannabis Regulation and Safety Act and by November 8, 2016, the voters of California passed Proposition 64 or the Medicinal and Adult-Use Cannabis Regulation and Safety Act (MAUCRSA). As a result, any person wishing to commercially cultivate cannabis lawfully in California and obtain a license from the California Department of Food and Agriculture must notify CDFW, and we now have staff in six Regions assigned to process those notifications.

CDFW has a unique role in regulating today's commercial cannabis cultivation, in that we have the opportunity to recommend measures designed to avoid and/or minimize impacts to fish and wildlife for every licensed cultivation site. As we progress beyond the first few years since MAUCRSA, we are likely to see changes to the way we regulate cannabis cultivation. Should future regulatory or legislative proposals consider weakening protections for fish and wildlife, we can consider the findings from the great work published here in this issue to make informed decisions and responses.

What does the future of regulating cannabis look like? I would like to explore more opportunities to collaborate with the cultivation community for voluntary restoration projects. We may have an opportunity to discuss safe harbor agreements or watershed-level restoration projects, and I look forward to that discussion.

[1] Johnson, N. 2019. American weed: a history of cannabis cultivation in the United States. EchoGéo 48. Available from: http://journals.openedition.org/echogeo/17650

[2] Brouwer, K. C., P. Case, R. Ramos, C. Magis-Rodriguez, J. Bucardo, T. L. Patterson, and S. A. Strathdee. 2006. Trends in Production, Trafficking and Consumption of Methamphetamine and Cocaine in Mexico. Substance Use & Misuse 41:707–727.

Introduction—continued

MOURAD W. GABRIEL, *United States Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, CA; University of California Davis, One Health Institute, Karen C. Drayer Wildlife Health Center, Davis, CA*

GRETA M. WENGERT, *Executive Director, Integral Ecology Research Center, Blue Lake, CA*

It is rewarding and galvanizing for us to contribute towards the Introduction of this journal's special issue focusing on cannabis cultivation in California. It is equally lifting to see the several innovative research papers that make up this issue coming on the heels of an unprecedented cannabis cultivation season where both unpermitted private as well as trespass public land cultivation appear to be unbridled in plant production and environmental damage. This is our ninth season of data collection on the vast environmental impacts of this activity, and collectively, we have documented over 650 cannabis cultivation sites. Nevertheless, this season is proving to be on par with our first data collection season in 2012 in terms of impacts to California's natural resources.

California is a truly unique landscape home to a rich and diverse amount of flora and fauna, combined with the highest number of endemic species in North America. Yet neither California's plants nor wildlife is immune to deleterious anthropogenic influences, making this state also home to the second-highest number of plant and wildlife species protected federally. We understand that not all 102 state-listed or 132 federally-listed wildlife species in California are in direct conflict with or impacted by the cultivation of cannabis. However, the question now faced by today's natural resource managers and conservationists in California is if and how the vast expansion of cannabis cultivation within the many ecoregions in the state will impact our sensitive wildlife species and their essential habitats.

What we do know is that several sensitive species in California are at risk of contamination or poisoning from the many types of toxicants used at cannabis cultivation sites in attempt to mitigate herbivory of cannabis plants by wildlife, curtail pilfering of food stores at trespass cultivation site camps, and reduce damage to cultivation infrastructure by wildlife (Gabriel et al. 2012, Franklin et al. 2019). We know that the consumption of water for permitted and illegal cultivation is immense and often exceeds what would be considered sustainable for many of the watersheds that support threatened and endangered salmonid populations and other sensitive aquatic species (Bauer et al. 2015). From a landscape perspective, it is also evident that the fragmentation caused by both permitted and illegal cultivation cumulatively results in significant habitat impacts associated with substantial increases in edge and deforestation (Wang et al. 2017). Yet the list of scarcely explored and unexplored effects of all aspects of cannabis cultivation remains substantial, and the field of study veritably remains "wide open" for those researchers willing to venture into this largely unknown, and sometimes dangerous realm.

This volume and the collection of papers within represent the current state of the science in the investigation of the environmental impacts of both legal and illicit cannabis cultivation in California. Though individual studies exploring the first documented effects of these sites on wildlife took place almost a decade ago, since then studies on this topic have been few and far between and only by a limited number of dedicated scientists. The clandestine nature of the black market component of the industry, the risks of working in often dangerous and uncertain conditions, and the unconditional need to work closely with other disciplines, most notably law enforcement agencies, make embarking on these studies precarious and often unclear. However, with the raised local, regional and national awareness on this issue gained in recent years, notably with the legalization of cannabis cultivation in 2018, came more interest, support, and the recognized need for understanding the full array of impacts that cultivation might have on California's ecosystems. The several papers within this journal not only provide primary research, but reviews and meta-analyses to continue the discussion on the environmental ramifications, best management practices, and creative approaches towards the conservation and sustainability of California's natural resources within this rapidly emerging issue.

Nevermore than now exists the need for scientists and researchers to intensify the collection of empirical data on this topic to develop the foundation for management and policy guidance. We must reflect on where we currently stand in comparison to just a few years ago. From the research published in 2012 (Gabriel et al. 2012) focusing on rodenticide impacts from public land cannabis cultivation to Bauer et al. (2015) highlighting cannabis proliferation and associated water use on private lands, these foundational papers set a precedent that scientific evidence, rather than anecdotal inference, should guide policy. This current set of research articles extends that scientific foundation supporting the development of strategies in these novel and ever-changing times in California cannabis policy.

[1] California Department of Fish and Game (CDFG). "Atlas of the Biodiversity of California." California Department of Fish and Game, Sacramento (2003).United States Fish and Wildlife Service (USFWS). "Environmental Conservation Online System (ECOS)" https://ecos.fws.gov/, Accessed August 20, 2020

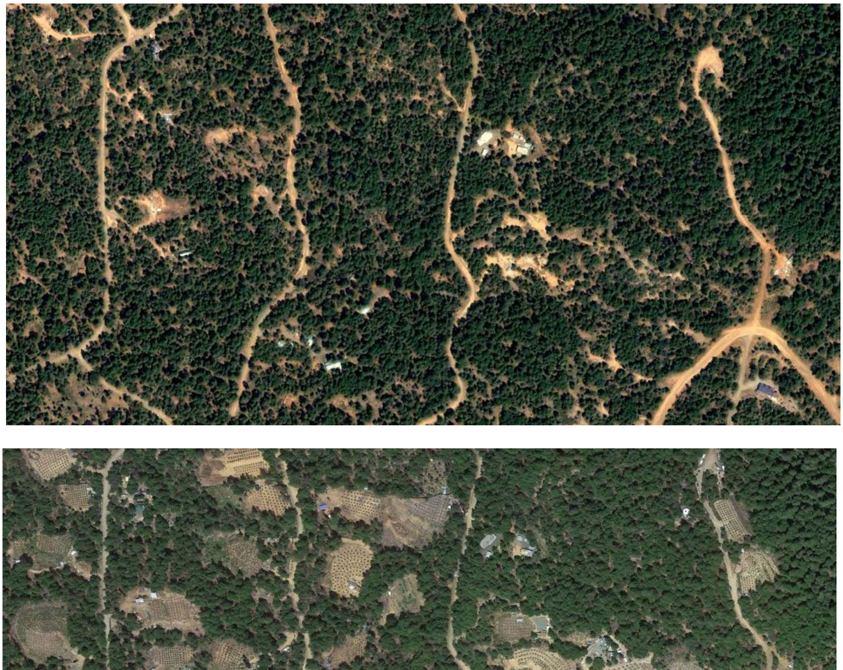
[2] Gabriel, M. W., L. W. Woods, R. Poppenga, R. A. Sweitzer, C. Thompson, S. M. Matthews, J. M. Higley, S. M. Keller, K. Purcell, R. H. Barrett, G. M. Wengert, B. N. Sacks, and de ana L. Clifford. 2012. Anticoagulant rodenticides on our public and community lands: Spatial distribution of exposure and poisoning of a rare forest carnivore. PLoS ONE 7:e40163.

[3] Franklin, A. B., P. C. Carlson, A. Rex, J. T. Rockweit, D. Garza, E. Culhane, S. F. Volker, R. J. Dusek, V. I. Shearn-Bochsler, M. W. Gabriel, and K. E. Horak. 2018. Grass is not always greener: rodenticide exposure of a threatened species near marijuana growing operations. BMC Research Notes 11:1–8.

[4] Bauer, S., J. Olson, A. Cockrill, M. Van Hattem, L. Miller, M. Tauzer, and G. Leppig. 2015. Impacts of surface water diversions for marijuana cultivation on aquatic habitat in four northwestern California watersheds. PLoS ONE 10:e0120016.

[5] Wang, I. J., J. C. Brenner, and V. Butsic. 2017. Cannabis, an emerging agricultural crop, leads to deforestation and fragmentation. Frontiers in Ecology and the Environment 15:495–501.

Eubank Creek, Humboldt County, CA, 2019. Example of stream morphology in the study area. Photo credit: Elijah Portugal, CDFW



Post Mountain, Trinity County, CA. Google Earth aerial images taken in the same location in 2007 and 2016 demonstrating land clearing for cannabis cultivation operations.

California Fish and Wildlife, Cannabis Special Issue; 13-29; 2020

Applied Science to Inform Management Efforts for Cannabis Cultivation, Humboldt, County, California

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Key words: aerial imagery, California Department of Fish and Wildlife, cannabis, environmental monitoring, remote sensing, watershed

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Like other forms of commercial agriculture, recent work has shown that land use practices associated with cannabis agriculture can pose a risk to aquatic and terrestrial habitat for threatened and endangered species (Bauer et al. 2015; Carah et al. 2015; Butsic and Brenner 2016; Butsic et al. 2018). Potential impacts from cannabis agriculture vary widely among different types of cultivators, ranging from illegal, clandestine public land trespass grows, privately owned non-compliant cannabis farms, and cumulative impacts associated with privately owned farms in the regulated market (Bodwitch et al. 2018; Schwab et al. 2019).

The focus of much recent work has been investigating impacts from illegal public land trespass grows (Gabriel et al. 2012, 2013, 2018; Thompson et al. 2014) or has not differentiated between private land cultivators based on their level of regulatory compliance (Butsic and Brenner 2016; Wang et al. 2017; Butsic et al. 2018). This paper focuses on the preliminary findings of a larger study examining the impacts of cannabis cultivation on private lands in remote, forested watersheds of northwestern California that have supported decades of illegal cultivation and include both compliant and non-compliant cannabis cultivators.

Clandestine public land trespass grows have been associated with poisoning of terrestrial wildlife (Gabriel et al. 2012, 2013, 2018; Thompson et al. 2014), and both clandestine and non-compliant private-land growers have been associated with illegal forest conversions and habitat fragmentation (Wang et al. 2017; Butsic et al. 2018) to support cannabis cultivation. Commercially available agricultural fertilizers and pesticides not unique to cannabis can degrade water quality and cause additional impacts to sensitive aquatic species (USEPA 1994; Alvarez et al. 2008a,b). Cumulative water diversions to support cannabis agriculture pose a high risk of reducing or seasonally eliminating critical aquatic habitat (Bauer et al. 2015; Dillis et al. 2019; Zipper et al. 2019).

Cannabis farms that are in compliance with current policies established by the State Water Resources Control Board (SWRCB) (2019) attempt to minimize impacts by following best management practices. These include measures that avoid sedimentation and erosion (e.g., minimum setbacks from riparian areas and streams), institute a forbearance period from surface water diversions for cannabis during the low-flow season, and reduce inputs of pathogens and toxicants into streams. With cannabis being a newly legalized industry within the state, policies are continuing to evolve and will presumably require continued assessment and monitoring to ensure that the potential impacts of legal commercial cultivation are minimized.

The cannabis industry (permitted and unpermitted production) nearly doubled in area under cultivation from 2012-2016 in Northern California (Butsic et al. 2018) and the quantity and magnitude of stream diversions associated with this expansion, as well as the potential for other forms of cumulative impacts, requires an objective, data-driven management response from the California Department of Fish and Wildlife (CDFW). To meet the mandate for environmental monitoring and management of the emerging cannabis industry, CDFW is developing the California Environmental Monitoring and Assessment Framework (CEMAF), a statewide monitoring framework to assess potential impacts to aquatic and terrestrial habitat and communities from all forms of outdoor, greenhouse, and mixed light cannabis cultivation and other land uses. To inform the development of CEMAF and to test assessment and analysis methods novel to CDFW, the Fisheries Branch and Water Branch within CDFW initiated a pilot study in the Headwaters Mattole River watershed (Hydrological Unit Code 12 (HUC12): 180101070202) in May 2018 that concluded in October 2019. This research note solely summarizes the methods, analysis and discussion of the cannabis cultivation site mapping portion of the 2018 pilot study. The findings presented here, and the findings of the larger overall pilot study informed the development of CEMAF but are not a product of CEMAF, which is still in development.

We employed high resolution aerial imagery and simple GIS analysis to identify cannabis cultivation sites and assessed their likelihood to impact aquatic and terrestrial habitat at three spatial scales (e.g., individual farm scale, watershed and entire study area). The three spatial scales were selected to meaningfully summarize results for land managers and to scale up the results at the farm scale to the watershed scale and larger to compare the potential for cumulative impacts. For the purposes of this note, we assumed that the five metrics below and related hypotheses would correspond to the likelihood of impacts to the aquatic, and to a lesser extent terrestrial, environment in the study area due to cannabis cultivation.

1) Farm Attributes: size, operation type and presence of a pond. We assumed with all else being equal, that a farm with a larger footprint of disturbance would be more likely to cause impacts to surrounding terrestrial and aquatic habitat than smaller ones. We also assumed that the demographics of farm owners would correspond to farm size and that may influence the ability of a given farm to join the regulated market. The operation type (e.g., outdoor or greenhouse) and presence of a pond were also identified as these features influence the amount of water extracted from the watershed to support cannabis production (Dillis et al. 2019) leading to potential impacts to instream flow.

2) Proximity to critical habitat for steelhead (*Oncorhynchus mykiss*). With all else being equal, we assumed that a farm was more likely to impact aquatic habitat if it was located in close proximity to designated critical habitat for steelhead.

3) Slope: proportion of sites located on steep slopes. The potential for erosion, sediment delivery and runoff containing toxicants from cultivation sites and roads is assumed to be relatively higher when the site or road occurs on steep slopes (e.g., >30%) compared to a low-gradient valley setting (Walling and Webb 1983; Liu et al. 2000; Verstraeten

and Poesen 2001). Excess fine sediment negatively influences growth, reproduction and mortality rates at all trophic levels in the aquatic environment with direct and indirect effects to freshwater fishes (Kemp et al. 2011). Excess sedimentation has been shown to be particularly detrimental to salmonid spawning through filling interstitial spaces in gravel, leading to a decrease in available oxygen in developing redds (Suttle et al. 2004; Sear et al. 2008; Kemp et al. 2011).

4) Compliance: proportion of sites with a temporary permit from California Department of Food and Agriculture (CDFA). We assumed that the likelihood for impacts from an unregulated farm was higher than one in the legal market that is attempting to minimize impacts by adhering to SWRCB cannabis policy land use practices.

5) Road Metrics: We analyzed four additional metrics associated with potential impacts from the road networks within the study watersheds. Long-standing empirical evidence shows that unpaved forest roads are a significant anthropogenic contributor of sediment to the aquatic environment at both the local and watershed scale (Reid and Dunne 1984; Bilby et al. 1989; Luce and Black 1999). The length, location, age, construction practices, amount of use, basin geology, and precipitation characteristics have all been shown to influence the amount of sediment generated from roads, but in general road networks cause a net increase in watershed scale sediment production (Reid and Dunne 1984; Bilby et al. 1989; Wemple et al. 2001). Though paved roads with adequate cut slopes and ditches typically produce 1% of the sediment yield produced by gravel roads under heavy use with all other factors being equal (Reid and Dunne 1984).

We assumed that attributes of the road network in the study watersheds would influence the potential for erosion, sediment delivery and increased peak flows where higher road densities, more road crossings of the stream network and more unpaved surfaces would have a higher likelihood of impacting aquatic and terrestrial habitat. Specifically, we assumed that the potential for road-related sedimentation was higher in locations where the road network was in close proximity to the stream network compared to areas where the road was further away.

METHODS

Study Area

The pilot study took place in coastal Northern California within five small, intermittent tributaries (mean drainage area = 11.45 ± SE 3.37 km^2) to the headwaters of the Mattole River (HUC12:180101070202; Figure 1). The Mattole watershed was selected because of the long history of clandestine cannabis cultivation in close proximity to high value aquatic and terrestrial habitat. Study streams in the Mattole River watershed were selected to possess a range of cannabis cultivation densities and willing landowners to provide access to their farms to meet the objectives of the larger study. All five study watersheds in the headwaters of the Mattole River were included in the aerial imagery/GIS analysis and gauged for hydrological assessment and three of them (Eubank, McKee, and Van Arken) also received biological assessment, though those results are in preparation for a separate technical report and are not presented here. Van Arken Creek was included as a reference watershed where no cannabis cultivation was present.

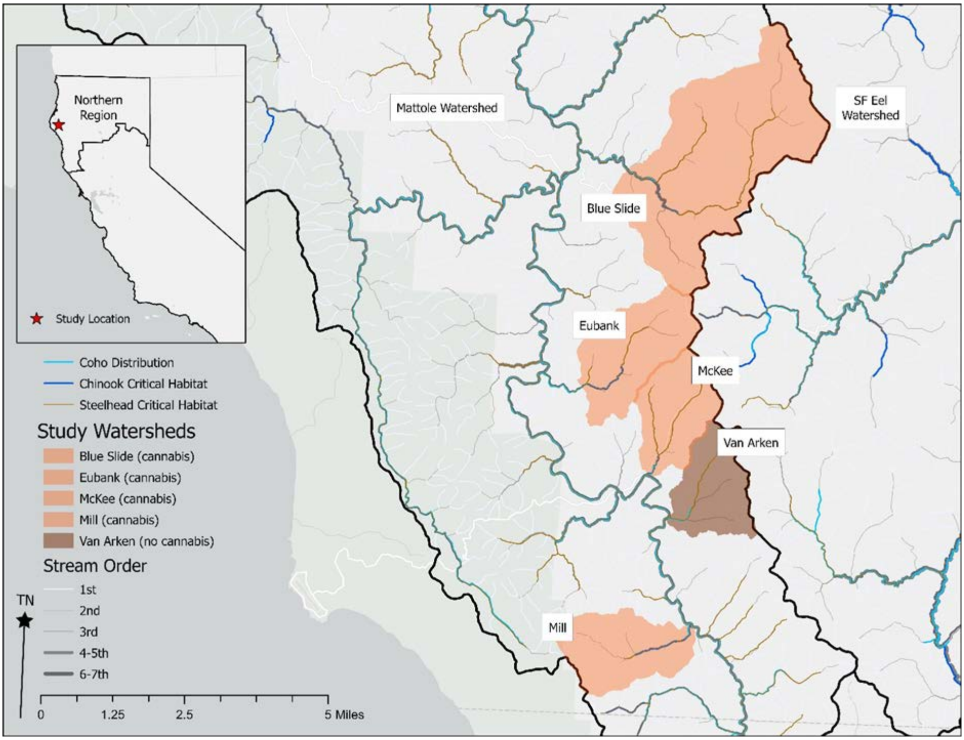

Figure 1. Map of watersheds included in the 2018 pilot study (n = 5). Peach fill indicates watersheds with known cannabis cultivation and brown fill indicates watersheds with no known cultivation.

Aerial Imagery Analysis/Geographic Information System (GIS)

Farm Attributes.—We manually digitized all cannabis cultivation sites (defined as individual greenhouses or outdoor gardens) identifiable from current aerial imagery within the five study watersheds by digitally tracing a polygon boundary around the footprint of each feature. Digitization included both compliant and non-compliant sites and all sites were attributed to a parcel or multiple continuous parcels which were then defined as a farm. A farm is defined as a discrete location that could contain multiple cultivation sites with greenhouses, outdoor gardens, and/or ponds. We followed the digitization methods developed in Bauer et al. (2015) and refined in Butsic and Brenner (2016) and in Butsic et al. (2018). We formalized this process by developing a guidance document for the manual digitization, storage and documentation of cannabis cultivation sites from aerial imagery (CDFW 2020 [mapping guidance doc]). We used both Google Earth (Google Maps 2018) and ESRI's ArcGIS mapping platforms (Esri 2018) to conduct the analysis.

In order to use the most current aerial imagery available to us within the study area, we primarily digitized in ArcGIS using ~ 30 cm resolution Digital Globe imagery (Digital Globe 2018) acquired on 25 April 2018 and 2 May 2018. We were concerned that we would not be able to adequately identify outdoor gardens using imagery acquired in April and May when outdoor plants may not be in place or are too small to individually identify. To address this, we cross-referenced the majority (~ 80%) of all sites identified from the April/May imagery with Digital Globe imagery acquired 9 October 2018, though this imagery did not cover all study watersheds. Thus 20% of sites could not be confirmed by the October imagery alone. This effort was to validate that greenhouses and outdoor gardens identified using the April/May imagery were under cannabis cultivation in October when outdoor cannabis plants attain their maximum size and are easier to identify. If any outdoor gardens identified using the April/May imagery did not show signs of cultivation from the October imagery, that site was deleted. Less than 10% of the outdoor gardens identified in April/May were deleted based on October imagery verification (i.e., 6 outdoor gardens were deleted of the 65 identified using April/May imagery). To address the 20% of sites that could not be verified with October 2018 imagery, we cross-referenced all cannabis sites identified in 2018 with Google Earth imagery from 28 May 2014 to gather more visual evidence that the sites identified from 2018 imagery were also under cannabis cultivation in 2014. This was not to determine a sites longevity but rather to provide more evidence that a site was growing cannabis and not a rural homestead with a large vegetable garden and greenhouse.

It is more difficult to detect outdoor gardens relative to greenhouses which are easily identifiable from aerial imagery. Though outdoor gardens do not appear to be the preferred operation type amongst cultivators in the study area with outdoor gardens only comprising 10.9% of all sites. It was impossible to confirm that all greenhouses identified were solely growing cannabis and not another greenhouse crop. There is evidence to suggest that it is unlikely that the greenhouses identified in the study watersheds are used for anything other than cannabis cultivation. For example, Butsic and Brenner (2016) compared the growth of greenhouses in Humboldt County from 2004–2014 to the growth of the nursery industry during the same period. They found the abundance of greenhouses increased 1900% while the value of nursery products in the county fell by 1.5% (Humboldt County 2015) indicating that greenhouses in Humboldt County were unlikely to be constructed and used for anything but cannabis. Though 2004–2012 is prior to the initiation of this study, the same logic applies in 2018, though it was not possible to verify the status of the non-cannabis nursery industry in Humboldt County from 2014–2018 because those data were not available. In many cases, it was possible to view historical imagery from sites that had been under cannabis cultivation for the last 5–10 years where we observed the transition from outdoor cannabis gardens to greenhouse cultivation.

Ponds associated with cannabis farms were also digitized within all study watersheds to use as input for a water extraction model (Dillis et al. 2019; CDFW, in prep.) though those results are not reported here.

1a) Farms with Ponds.—Dillis et al. (2019) showed that most cultivators in 2017 (n = 608) lacked the amount of storage (e.g., water tanks, bladders, and ponds) needed to meet late summer water demand unless they had a seasonal water source with a pond present. A seasonal water source was defined as rainwater catchment, springs, or surface water. If cultivators had a seasonal water source with a pond for storage, they were predicted to have a positive water storage balance for cultivation sites of up to nearly 0.4 ha (Dillis et al. 2019). As such, we identified all ponds visible during aerial imagery analysis of the study region.

1) Proximity to Critical Habitat

We calculated the distance of each cultivation site to NOAA/NMFS designated critical habitat for steelhead (National Marine Fisheries Service 2005) using the 'Near' tool in ArcGIS Desktop Version 10.5.1. The 'Near' tool provides the shortest geodesic distance and additional proximity information between each cultivation site relative to steelhead critical habitat. We summarized these data at the watershed scale and assumed that the likelihood of impacts to critical steelhead habitat from cannabis land use was relatively higher if the site was located within 45.7 m of critical habitat, which is the riparian setback distance required by the SWRCB (2019) Cannabis Policy of >45.7 m from perennial (Class I) watercourses.

2) Slope

To provide context of the study area's steep mountainous setting, slope rasters were generated from 10 m resolution Digital Elevation Models (DEMs) of each study watershed in ArcGIS to assess 1) mean watershed slope and 2) if each cultivation site occurred on a steep slope (>30%).

3) Compliance

For the purposes of this analysis we considered a farm in regulatory compliance if they possessed a temporary or annual cultivation license from CDFA. We assessed the proportion of sites that had obtained a temporary permit from CDFA using their license data for each parcel in the study watersheds. We then compared the license data with the locations of all cannabis cultivation sites identified from aerial imagery analysis to generate the % compliant metric at the watershed scale.

4) Road Metrics

We used two different methods to quantify road crossing metrics. The first metric examined the mean number of road crossings of NOAA/NMFS designated critical habitat for steelhead combined with locations where the road network was in close proximity (<15.2 m) to critical habitat (hereafter, "critical habitat roads metric"). The second metric did not consider critical habitat designation but included the number of road crossings of the perennial and intermittent stream network combined with the number of locations where the road came in very close proximity to the stream network (<15.2 m; hereafter "all roads metric"). We included the "all roads metric" because significant portions of the stream networks within the study watersheds extend upstream beyond the areas designated as critical habitat for steelhead and are in close proximity to the road network. These upstream portions of the stream network are hydrologically connected to the downstream critical habitat and are vulnerable to road-related sedimentation transported downstream into critical habitat.

We obtained the road network spatial data from Humboldt Counties GIS portal which is updated as needed (Humboldt County Building & Planning 2019). It was beyond the scope of this study to perform detailed road inventories required to generate quantitative predictions of sediment yield from the existing road network in the study watersheds. Instead, we generated four simple GIS metrics to assess the relative likelihood of road-related sedimentation and erosion impacts to sensitive aquatic habitat. These metrics were:

1) Road density (km/km^2),

2) Number of road crossings and locations where the road was <15.2 m from designated critical habitat for steelhead,

3) Number of road crossings and locations where the road was located <15.2 m from the perennial and intermittent National Hydrography Dataset (NHD) stream layer (not designated critical habitat),

4) Proportion of road network that is paved.

RESULTS

1) Farm Attributes

Within the study area at the regional scale, the total area under cultivation in 2018 was low (<1% of total drainage area; Table 1). There were 18.5 ha of cannabis cultivation in all four study sub-watersheds containing cannabis out of a total combined drainage area of 3,8401 ha (0.48% total drainage area). At the watershed scale, the total area under cannabis cultivation was also low (<1% total drainage area; Table 1). Of the watersheds with cannabis cultivation, Eubank Creek had the highest proportion of drainage area under cultivation (0.84%) and Mill Creek had the lowest (0.17%). One hundred and twenty-four farms were identified within the study area with a mean farm size of 0.12 ha.

1a) Farms with Ponds.—At the regional scale 11.4% of farms had a pond present. At the watershed scale, the proportion of ponds ranged from 6.7% (McKee) to 21.4% (Blue Slide; Table 1).

2) Proximity to Critical Habitat

The mean distance (± SE) of cultivation sites to critical habitat for steelhead across all study watersheds was 389.2 ± 65.2 m (Table 1). This is a substantially greater distance than the riparian setback distance of >45.7 m from perennial watercourses. For three of the four study watersheds containing cannabis, the mean distance to steelhead critical habitat was near the regional mean, with the exception of McKee Creek, which was considerably lower (mean distance = 175.3 ± 19.5 m; Table 1) when compared qualitatively. The proportion of sites <45.7 m from critical habitat for steelhead exhibited a similar trend with three of the four study watersheds containing cannabis having relatively low proportions (0-7.9%), while McKee Creek had the highest proportion with 25.8% of sites <45.7 m.

3) Slope

Though the total footprint of cultivation within the study watersheds was low, the location of farms relative to steep slopes (i.e., >30% slope) was moderate with 29.4% (n = 115) of all cultivation sites occurring on steep slopes. At the watershed scale, the proportion of sites considered steep varied from 19% (Eubank) to 36.2% (McKee). Mean basin slope for all watersheds is high (36.1% ± 1.39%). In comparison to the mean basin slope of each study watershed, cannabis sites tended to be located in less steep locations than the average slope conditions available within the watershed. Mckee Creek is an exception to this where mean basin slope and the proportion of farms considered steep were approximately equal.

Table 1. Summary of manual mapping results from study watersheds containing cannabis cultivation. (Note: Van Arken is not included because there is no cannabis cultivation in the watershed.)

Watershed	Drainage Area (km^2)	# Cannabis Farms	# Greenhouses (sites)	# Outdoor Gardens (sites)	Total Area Cultivated (ha)	% Drainage Area Cultivated	Mean Farm Area (ha)	% Farms with Ponds	Cultivation Site mean dist. to Steelhead Critical Habitat (m)	% Cultivation Sites <45.7 m from Steelhead Critical Habitat	% Cultiva-tion Sites >30% Slope	% Compliant
Eubank	6.2	27	146	13	5.2	0.84	0.19 ± .04	7.7	472 ± 22	1.3	19.0	63.0
McKee	5.4	16	55	10	1.9	0.36	0.12 ± .02	6.7	175 ± 20	25.8	36.2	35.3
Blue Slide	20.7	71	257	33	10.4	0.50	1.21 ± .02	21.4	397 ± 20	8.3	27.1	26.3
Mill	6.0	10	35	2	1.1	0.17	0.11 ± .02	10.0	513 ± 45	0.0	35.3	10.0
All Sites†	9.6 ±3.1×	124*	482*	59*	18.5*	0.50×	0.15 ± .01×	11.4×	389 ± 65×	8.8×	29.40×	33.6×

†Value in this row include both sums (denoted with *) and means ± standard error (denoted with ×).

4) Compliance

At the regional scale, 33.6% of all sites possessed either a temporary or annual license from CDFA and were considered compliant for this analysis (Table 1). At the watershed scale, the proportion of compliant sites in 2018 ranged from 10% (Mill) to 63% (Eubank; Table 1).

5) Road Metrics

Road Density.—At the regional scale, road density was 2.8 km/km^2 (Table 2) with notable variability between study watersheds. Van Arken, the reference watershed had approximately 2 times higher road density compared to the regional mean.

Road Crossings.—At the regional scale, the critical habitat roads metric was 2.2 ± 0.6 km/km^2 (Table 2). At the watershed scale, this metric ranged from 1 (Eubank) to 6 (Van Arken). We found that the all roads metric was higher than the critical habitat roads metrics in all study watersheds, with the exception of McKee Creek, where these metrics were equal to one another (Table 2). The regional mean for the all roads metric was 5.3 ± 2.0 km/km^2, which is greater than two times the value of the critical habitat roads metric.

Proportion Paved.— At the regional scale, the mean proportion of paved roads within the study watersheds was 10.9% ± 6.4% indicating that the vast majority of all roads in the study watersheds are unpaved. The regional mean is primarily driven by the high proportion of paved roads in McKee Creek (37.8%) and moderate proportion in Eubank (13.6%), while the rest of the study watersheds were essentially unpaved (0-2.8% paved).

Table 2. Summary of GIS derived metrics associated with potential impacts from the existing road network study watersheds.

Watershed	Road Density (km/km^2)	Road Crossings and Roads within 15.2 m of Critical Steelhead Habitat	Road Crossings and Roads within 15.2 of Stream Network (Perennial and Intermittent)	Proportion Paved
Eubank	2.49	1	2	13.6
McKee	2.24	5	5	37.8
Blue Slide	2.54	5	21	2.8
Mill	1.26	1	3	0.0
Van Arken	5.44	6	12	0.5
All Sites (mean)	2.79 ± 0.63	3.6 ± 0.96	8.6 ± 3	10.9 ± 6.4

DISCUSSION

Aerial imagery analysis of cannabis sites combined with simple GIS metrics represents a tractable methodology to assess relative risk of impacts from cannabis cultivation land-use to aquatic and terrestrial habitat in the study area. However, it was beyond the scope of this study to quantitatively rank each of the metrics in terms of their ability to describe impairment due to cannabis cultivation. The total footprint of cannabis cultivation within the study watersheds was low (<1% of total drainage area; Table 1) and average farm size was low (0.12 ha) with farms not generally located in close proximity to designated critical habitat (mean = 389.2 ± 65.2 m) and with relatively high levels of regulatory compliance (33.6%). This indicates that at the regional scale, the potential impacts from cannabis cultivation in the study area may be low. When viewed at the scale of individual study watersheds or individual farms, the potential for impacts is more variable. The location of many farms is problematic due to the proximity to designated critical habitat (Mckee Creek site mean distance = 175.3 ± 19.5 m), the steep headwaters setting of the study watersheds (mean basin slope = 36.1% ± SE 1.39%), and the presence of unpaved road networks with relatively high road density (2.8 km/km^2) that occur on steep slopes and cross-designated critical habitat.

1) Farm Attributes

The mean farm size in the study area was >2 times the mean farm size reported in Butsic et al. (2018). There are a few likely reasons for this disparity. Butsic et al. (2018) assessed a much larger area with far more farms compared to this study (n = 5906 and 124, respectively). Consequently, they were better able to capture the full range of variability in farm size throughout the cannabis producing regions in Northern California. The difference in farm size may also reflect regional differences in the cannabis industry. The Mattole River Watershed has been a cannabis cultivation hotspot for decades and established, multi-generational cultivators are more likely to have the resources to navigate the regulatory process and sustain larger farms relative to the cultivators that have entered the industry recently during the unregulated “Green Rush” of 2012-2016 (Butsic et al. 2018). Additionally, we were able to use slightly more recent imagery than Butsic et al (2016, 2018) and the trend in increasing farm size he documented would have likely continued in the few years between the studies.

The dominance of greenhouses in our study region (90% of all sites are greenhouses) is much higher than Butsic and Brenner’s (2016) findings based on aerial imagery analysis from 2012-2013. They found the proportion of greenhouses to outdoor gardens was approximately equal (54% greenhouses). The discrepancy could be a matter of scale of the studies as mentioned previously, and/or also reflect a difference in the demographics of cultivators with more established growers in the Mattole watershed favoring greenhouses. It could also reflect broader changes in the cannabis industry since 2013 where there was a transition to relatively more greenhouse production from a previously even distribution of outdoor gardens and greenhouses. This is consistent with Butsic et al. (2018) where they documented a 248% increase in the amount of plants grown in greenhouses from 2012-2016 relative to total plant increase (greenhouses and outdoor gardens) of 183%. Greenhouse production allows for a longer growing season, more harvests per year, and potentially higher yields compared to outdoor gardens.

1a) Farms with Ponds.— The vast majority (88.6%) of cultivators in the study area in 2018 likely did not have enough storage to meet late summer water demands as evidenced by the lack of ponds. Consequently, it is also likely that water extraction for cannabis occurred during the critical low flow period of July through October 2018. We also assume that well use occurred during the low flow period though it was beyond the scope of this study to determine the total number of well users and the magnitude and frequency of groundwater diversion. We also did not examine the level of hydrologic connection between groundwater and surface water in the study watersheds. The magnitude and intensity of water extraction during the low flow period cannot be verified with absolute certainty because we could not inventory all water sources or storage infrastructure for all cultivators but our findings are consistent with Dillis et al. (2019) who showed that most cultivators (n = 608) enrolled with the North Coast Regional Water Quality Control Board lacked the amount of storage (e.g., from water tanks, bladders, and ponds) needed to meet late summer water demand unless they had a seasonal water source with a pond present.

The difference in the location and density of ponds amongst study watersheds may relate to characteristics of the underlying lithology which has been shown to influence a watersheds ability to store water as groundwater in the winter and slowly release that water as baseflow in the late summer (Davenport et al. 2002, Lovill et al. 2018). For example, Blue Slide creek had the highest proportion of sites with ponds (21.4%) compared to the regional average (11.4%) and is underlain by a mélange rock type with lithology associated with low levels of groundwater storage. With relatively less groundwater available in late summer in Blue Slide Creek relative to other study watersheds underlain by rock types that can support higher levels of groundwater storage (Hahm et al. 2019), a pond is necessary to store surface and groundwater collected in the wet winter months to meet late summer plant demand.

2) Proximity

The mean distance of cultivation sites to critical habitat for steelhead across all study watersheds was 8 times greater distance than the 45.7 m riparian setbacks required by SWRCB Cannabis Policy (2019) for perennial (Class I) watercourses. An exception to this was McKee Creek, possessing the highest proportion of sites (25%; n = 17) located <45.7 m from critical habitat for steelhead. Four out of five study watersheds possessed a very similar proportion of available steelhead habitat within the drainage network. Again, this is evidence of the variability of potential impacts from cannabis cultivation when viewed at multiple spatial scales.

3) Slope

High mean basin slope of the study watersheds coupled with underlying lithology that is highly erosive (Davenport et al. 2002) in a climate with high intensity winter precipitation events creates a combined physiographic setting that is naturally prone to mass wasting and transport of sediment into stream networks. In addition to the physiographic setting, the study area experienced decades of anthropogenic impacts to the watershed-scale hydrologic and sediment routing processes from large-scale forest conversions and road development primarily to support commercial timber extraction prior to large-scale cannabis cultivation. This resulted in a landscape that is vulnerable to additional anthropogenic impacts from cannabis cultivation, particularly in steep locations. When roads and cannabis farms are located on steep slopes or land conversions to support cannabis, there is an elevated risk of sediment-related impacts to aquatic habitat in the study area.

The distance to the stream network also influences the likelihood of impacts to aquatic habitat from mass wasting or toxicant runoff initiated from a cannabis farm located on a steep slope. The sites that are on steep slopes (>30%) and located within close proximity to the stream network (<45.7 m) are at the highest risk for impacting nearby aquatic and riparian habitat though the proportion of sites that met that criteria was very low (4.1%; n = 22). In general, it appears that cultivators tend to locate their farms on less steep locations relative to available slope conditions in the watersheds.

4) Compliance

The trends in compliance we identified from aerial imagery analysis revealed the different demographics of cannabis cultivators in this region. Generally, cannabis cultivators in the headwaters of the Mattole have a much higher level of regulatory compliance than the statewide mean (33.6% of sites = compliant). This is approximately three times higher than anecdotal estimates from state cannabis regulators of approximately 10% compliance among the total cannabis industry in California and 30 times higher than a recent report from the California Growers Association (California Growers Association 2018) asserting that 1% of the state's cannabis cultivators have joined the legal market.

The watershed with the highest level of compliance, Eubank Creek (63%), also had the largest mean farm size (0.19 ± 0.04 ha) and the largest proportion of the watershed under cultivation (0.84%). These combined metrics may reveal a difference in the demographics of cultivators in the region where the largest farms tend to be owned by cultivators with more financial resources and motivation to join the regulated market (Polson and Petersen-Rockney 2019, Schwab et al. 2019, Wilson et al. 2019). This finding is consistent with Butsic et al. (2018) who found that large farms were less likely to be abandoned than small farms and that smaller farms are less likely to join the regulated market. Specifically, Butsic et al. (2018) found that farm abandonment between 2012-2016 was best predicted by farm size, with smaller farms (i.e., ≤50 plants) twice as likely to be abandoned relative to large farms (i.e., ≥200 plants).

The variability in rates of compliance amongst study watersheds was likely driven by a combination of physical and demographic factors. For example, McKee Creek had the highest proportion of farms on steep slopes (36.2%) and the highest proportion of farms within 45.7 m of critical steelhead habitat (25.8%) which is three times higher than the regional mean. These attributes make the permitting process more difficult for cultivators to become compliant in McKee Creek relative to Eubank Creek and the difference in the rates of compliance between the watersheds is apparent (Eubank = 63%; McKee = 35.3%). Eubank Creek had the largest mean farm size (0.19 ± 0.04 ha) and the lowest proportion of sites located on steep slopes (19.0%), and the second lowest proportion of sites within 45.7 m of critical habitat for steelhead (1.3%; Table 1). These factors increase the likelihood of obtaining a permit because the sites are considered lower risk and do not require sediment and erosion plans by the SWRCB's (2018) Cannabis Policy.

5) Road Metrics

Road density within the study watersheds is >4 times the road density considered an 'acceptable' level (0.6-0.7 km/km^2) to sustain a naturally functioning landscape that supports large terrestrial predators (Forman and Alexander 1998). This threshold has been used in recent studies examining impacts of roads on terrestrial organisms (e.g., Cai et al. 2013, Boulanger et al. 2014). The National Research Council (2005) identified a threshold range of road density between 2.0-3.0 km/km^2 and at road densities greater than the threshold, alterations to the runoff regime and flow routing processes at a watershed scale are pervasive and peak flows typically increase (National Research Council 2005). Increased peak flows often lead to a decrease in instream habitat quality and quantity (Poff et al. 1997). All study watersheds except Mill Creek, had road densities equal to or greater than 2.0 km/km^2, indicating that peak flows have likely been altered from baseline conditions in three of the four study watersheds.

One major caveat of the current study is that the highest road density and number of crossings within 15.2 m of critical steelhead habitat occurred in the reference watershed, Van Arken Creek. This reflects the recent history of timber production as the dominant land use in that watershed as opposed to cannabis cultivation in the others. In Van Arken Creek, legacy impacts to hydrological and sediment routing processes from commercial timber production are likely still impacting the quantity and quality of instream habitat (CDFW, in prep).

Other studies have shown that the amount of sediment yield to the stream network associated with road crossings is widely variable based on the construction methods, surface type, and condition (Wemple et al. 2001). As such it was beyond the scope of this study to rank the likelihood of impacts from each road crossing. Despite this, the total number of crossings may still be a reasonable proxy to assess the relative likelihood of increased sediment yield and erosion potential at the watershed scale.

CONCLUSIONS/RECOMENDATIONS

Aerial imagery analysis to identify cannabis cultivation sites combined with simple GIS metrics associated with: 1) farm attributes, 2) proximity to critical habitat, 3) site and watershed slope, 4) regulatory compliance, and 5) attributes of the road networks provide the basis for developing a screening tool for rapidly assessing the relative risk of impacts to the aquatic and terrestrial environment from large-scale cannabis cultivation without the need for extensive field visits. The GIS metrics presented here are not comprehensive, and more robust metrics could be developed and employed statewide that assess the risk of cannabis-related impacts associated with natural intrinsic watershed characteristics (e.g., geologic setting as it relates to groundwater availability, climate/hydrologic setting, vegetation, presence of species of special concern), and anthropogenic drivers like recent land conversions for cannabis or the total number of surface water diverters and well users in a given watershed. With repeat cannabis site mapping in the same location, a land conversion metric could be generated that assesses the relative risk of cannabis impacts due to the history and magnitude of land conversions for cannabis cultivation. Similarly, hydrologic metrics could be generated to assess the risk for dewatering a given watershed based on the hydrologic setting (i.e., estimates of unimpaired flow) relative to total water users employing a water budgeting approach (Zipper et al. 2019). The metrics described here could also be refined. For example, our %compliant metric could be bolstered to account for all state and local permits required in a given county and not solely based on CDFA licenses. The road-related sedimentation metrics could also be significantly bolstered by rapid, field-based road assessments to investigate the condition of road crossings and unpaved roads close to the stream identified in the GIS analysis.

In the absence of more detailed metrics or some level of field validation, we have shown that potential impacts from cannabis cultivation in our study area are variable based on the spatial scale of inquiry, the actual farm locations relative to sensitive habitat, and intrinsic watershed characteristics (e.g., steep slopes) that make a given farm more likely to impact the aquatic environment. The use of geospatial information to identify land use types and analyze associated impacts is well-established and here, we use geospatial data to explore metrics related to cannabis cultivation. The methods we describe will be used to help with the development of CEMAF, a robust statewide monitoring framework to help CDFW scientists assess the impacts of cannabis cultivation on aquatic and terrestrial ecosystems.

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Great horned owl (*Bubo virginianus*) on an artificial perch at a cannabis cultivation in southern Humboldt County. Owls are an excellent natural rodent-control agent and providing perches and nest boxes for them can increase their presence on cultivation sites and reduce the need for rodenticides. Photo Credit: Ryan Mathis, CDFW

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Pesticides in California: their potential impacts on wildlife resources and their use in permitted cannabis cultivation

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The agricultural industry, including commercial cannabis cultivators, often relies on rodenticides and insecticides to help minimize damage from wildlife and insect pest species. Many of the most toxic pesticides are listed as California restricted materials, meaning they can only be purchased and used by certified applicators under a permit from a County Agricultural Commissioner. Despite the permit requirement and other restrictions, exposure of non-target wildlife to pesticides continues to occur throughout California. Non-target wildlife may be directly exposed through ingestion, inhalation, or dermal contact or secondarily exposed through ingestion of contaminated or poisoned prey. Exposure to pesticides can be lethal, or it can cause sublethal effects that impact species' immunology, reproduction, thermoregulation, morphology, and behavior. To date, information pertaining to pesticides is spread among disparate resources. Our review paper aims to synthesize a subset of this information. We provide an overview of insecticides and rodenticides and explore the potential effects that these pesticides may have on non-target wildlife species. We then outline current regulations regarding the use of these pesticides in cannabis cultivation, one of the fastest growing agricultural commodities in California.

Key words: acute poisoning, cannabis, insecticide, pesticide, regulations, rodenticide, sublethal effects, wildlife

The agricultural industry often relies on pesticides to control wildlife and insect pests that damage plants by foraging on them or using them as nesting material (NDIC 2007). Following California Food and Agricultural Code section 12753(b), we define pesticides as any substance, or mixture of substances, which is intended to be used for defoliating plants, regulating plant growth, or for preventing, destroying, repelling, or mitigating any pest species. Thus, the term pesticide is an overarching term that encompasses, for example, rodenticides, herbicides, insecticides, fungicides, acaricides, and nematicides. In this review, we focus on insecticides and rodenticides because they tend to be more acutely toxic to non-target wildlife species than other types of pesticides.

Pesticides are an issue of conservation concern because they can negatively impact species that are not the focus of pest control activities such as non-target wildlife (Table 1). Non-target wildlife may be directly exposed through ingestion, inhalation, or dermal contact or indirectly exposed through the ingestion of contaminated or poisoned prey (Berny 2007). An additional effect may be diminished prey resources for species that depend on insects or rodents as a food source (Hallmann et al. 2014). Exposure may happen in the area where pesticides were applied, or in surrounding soils, ground water, or surface waters given pesticides can move via spray drift, surface runoff, soil erosion, leaching, or irrigation return flows (Pimentel 2005; Baldwin et al. 2009). The off-target movement of pesticides has resulted in over 10% of the watersheds within California's North Central Coast, South Central Coast, Middle Sacramento, South Sacramento, and San Joaquin containing streams that are impaired by pesticides, where impairment is defined as surface waters that contain pollutants at levels that exceed protected water quality standards (SWRCB 2018). More than 50 pesticides and pesticide breakdown products were detected in the San Joaquin and Sacramento water basins alone (Baldwin et al. 2009; SWRCB 2018).

Currently, information pertaining to pesticides is spread among disparate scientific, management, and regulatory resources. The goal of our review is to synthesize a subset of this information, with the specific objectives of 1) providing an overview of insecticides and rodenticides, two groups of pesticides likely to impact wildlife, 2) describing the acute and sublethal effects of insecticide and rodenticide exposure on non-target wildlife, and 3) outlining current regulations regarding pesticide use on permitted cannabis cultivation sites. We focus on cannabis cultivation, specifically, because it is one of the fastest growing agricultural commodities in California. Further, we aim to distinguish permitted cannabis cultivation, which has stringent pesticide use regulations, from illegal cannabis cultivation, where the use of California and federally restricted pesticides and numerous ensuing environmental impacts have been documented (Gabriel et al. 2012, 2015, 2018).

PESTICIDE GROUPS

Insecticides

Among the most toxic pesticides are organophosphate and carbamate insecticides, known as the anticholinesterase pesticides (Fleischli et al. 2004). These insecticides function by inhibiting acetylcholinesterase (AChE), an enzyme essential to the functioning of the nervous system (Grue et al. 1997; Baldwin et al. 2009). When AChE accumulates in the nervous system due to inhibition by these chemical families, there is uninterrupted stimulation, loss of energy from signal receptors, and eventually, paralysis of respiratory muscles, asphyxiation, and death (Fleischli et al. 2004). Additionally, sublethal exposure to organophosphate and carbamate insecticides can cause short-term hypothermia, decreases in food consumption, weight loss, impaired vision, and altered sexual behavior, with effects tending to be especially acute in birds (Grue et al. 1997). Many insecticides made of

Table 1. Some of the non-target mammal and bird species documented to have pesticide residues in their systems or to have died from acute or secondary poisoning from pesticides. The table is based on information in Stone et al. (1999), Hosea (2000), Erickson and Urban (2004), Brakes and Smith (2005), Lima and Salmon (2010), Elliott et al. (2014), and CDPR (2018).

Mammals		Birds	
Common name	Species name	Common name	Species name
American badger	*Taxidea taxus*	American crow	*Corvus brachyrhynchos*
Bank voles	*Clethrionomys glareous*	American kestrel	*Falco sparverius*
Bobcat	*Lynx rufus*	Bald eagle	*Haliaeetus leucocephalus*
Coyote	*Canis latrans*	Barn owl	*Tyto alba*
Eastern chipmunk	*Tamias striatus*	Barred owl	*Strix varia*
Ermine	*Mustela erminea*	Burrowing owl	*Athene cooperii*
Field voles	*Microtus agrestis*	Common raven	*Corvus corax*
Fisher	*Pekania pennanti*	Cooper's hawk	*Accipiter cooperii*
Gray fox	*Urocyon cinereoargenteus*	Eastern screech owl	*Megascops asio*
Gray squirrel	*Sciurus carolinensis*	Golden eagle	*Aquila chrysaetos*
Heermann's kangaroo rat	*Dipodomys heermanni*	Great horned owl	*Bubo virginianus*
Long-tailed weasel	*Mustela frenata*	Long-eared owl	*Asio otus*
Mountain lion	*Puma concolor*	Northern harrier	*Circus cyaneus*
Opossum	*Didelphis virginiana*	Northern spotted owl	*Strix occidentalis caurina*
Racoon	*Procyon lotor*	Peregrine falcon	*Falco peregrinus*
Red fox	*Vulpes vulpes*	Red-shouldered hawk	*Buteo lineatus*
San Joaquin kit fox	*Vulpes macrotis mutica*	Red-tailed hawk	*Buteo jamaicensis*
Striped skunk	*Mephitis mephitis*	Red-winged blackbird	*Agelaius phoeniceus*
Vole	*Microtus* spp.	Sharp shinned hawk	*Athene striatus*
White-tailed deer	*Odocoileus virginianus*	Snowy owl	*Bubo scandiacus*
Woodmice	*Apodemus sylvaticus*	Song sparrow	*Melospiza melodia*
		Swainson's hawk	*Buteo swainsonii*
		Turkey	*Meleagris gallopavo*
		Turkey vulture	*Cathartes aura*

these chemical compounds are listed as California restricted materials (California Code of Regulations Title 3, Division 6, §6400) that can only be purchased and applied by certified applicators under permit from a county commissioner (Table 2). The use of organophosphates and carbamates has decreased substantially in the US in recent decades as newer insecticides have gained favor.

Table 2. California restricted materials (refer to California Code of Regulations Title 3, Division 6, §6400 for details) that can only be purchased and applied by certified applicators under permit from a county commissioner and their corresponding type and substance group.

Pesticide	Pesticide type	Substance group
Acrolein*	Herbicide	Aldehyde
Aldicarb	Acaricide / Insecticide / Nematicide	Carbamate
Aluminum phosphide	Insecticide / Rodenticide	Inorganic compound
4-aminopyridine	Acaricide / Avicide	unclassified
Azinphos-methyl	Acaricide / Insecticide	Organophosphate
Brodifacoum	Rodenticide	Hydrocoumarin
Bromadiolone	Rodenticide	Coumarin anticoagulant
Calcium cyanide	Fungicide / Herbicide	Fertilizer
Carbaryl*	Insecticide	Carbamate
Carbofuran	Acaricide / Insecticide / Nematicide	Carbamate
Chloropicrin	Insecticide / Nematicide	Unclassified
Chlorpyrifos*	Insecticide	Organophosphate
3-Chloro-p-toluidine hydrochloride	Avicide	
Dazomet*	Fungicide / Herbicide / Insecticide	Carbamate
Dicamba*	Herbicide	Benzoic acid
2,4-Dichlorophenoxyacetic acid*	Herbicide	Alkylchlorophenoxy
2,4-Dichlorophenoxybutyric acid	Herbicide	
2,4-Dichlorophenoxy propionic acid*	Herbicide	Aryloxyalkanoic acid
1,3-Dichloropropene	Bactericide / Nematicide	Halogenated hydrocarbon
Difenacoum	Rodenticide	Hydroxycoumarin
Difethialone	Rodenticide	Coumarin anticoagulant
Disulfoton*	Acaricide / Insecticide	Organophosphate
Endosulfan*	Acaricide / Insecticide	Organophosphate
Ethoprop*	Insecticide / Nematicide	Organophosphate

Table 2. continued.

Pesticide	Pesticide type	Substance group
Fenamiphos	Nematicide	Organophosphate
Lindane*	Acaricide / Insecticide	Organochlorine
Magnesium phosphide	Insecticide / Rodenticide	Inorganic compound
Metam sodium	Fungicide / Herbicide / Insecticide / Nematicide	Carbamate
Methamidophos	Acaricide / Insecticide	Organophosphate
Methidathion	Insecticide	Organophosphate
Methomyl*	Acaricide / Insecticide	Carbamate
Methyl bromide	Acaricide / Insecticide	Organophosphate
Methyl iodide	Insecticide / Nematicide	Alkyl iodide
2-Methyl-4-Chlorophenoxyacetic acid*	Herbicide	Auxin
Methyl isothiocyanate	Fungicide / Herbicide / Insecticide / Nematicide	Unclassified
Mevinphos	Acaricide / Insecticide	Organophosphate
Molinate	Herbicide	Thiocarbamate
Oxydemeton-methyl	Insecticide	Organophosphate
Paraquat	Herbicide	Bipyridylium
Parathion-methyl	Insecticide	Organophosphate
Phorate	Acaricide / Insecticide / Nematicide	Organophosphate
Phosphine gas	Insecticide	Unclassified
Potassium n-methyldithiocarbamate*		
Propanil	Herbicide	Anilide
Sodium cyanide		Inorganic compound
Sodium fluoroacetate	Rodenticide	Organohalide
Sodium tetrathiocarbonate		
Strychnine*	Avicide / Rodenticide	Plant derived
Sulfotep	Acaricide / Insecticide	Organophosphate
Sulfuryl fluoride	Fungicide / Insecticide / Rodenticide	Inorganic compound
Thiobencarb	Herbicide	Thiocarbamate
Tribufos	Herbicide	Organophosphate
Tributyltin	Fungicide	Organometal
Zinc phosphide*	Rodenticide	Unclassified*

* Includes exceptions from restriction

Synthetic pyrethroid insecticides were developed in the 1970s and began replacing organophosphates and carbamates in the 1990s due to their lower toxicity to mammals and birds (Bradbury and Coats 1989; Casida and Quistad 1998). Pyrethroids alter insect neural membranes, which disrupts electrical signaling in the nervous system and ultimately leads to paralysis and death (Soderlund 2010). While pyrethroids are less toxic to mammals and birds than anticholinesterase pesticides, they are still highly toxic to fish and invertebrates when introduced in aquatic habitats (Casida and Quistad 1998; Soderlund 2010). Further, they have a high potential to contaminate downstream habitats given their low solubility in water, high absorption coefficient, and stability in sediment (Bradbury and Coats 1989). Pyrethroids are widely used in agriculture and in structural pest control and can be applied by both pest control professionals and non-professionals such as homeowners.

Neonicotinoids were introduced in 1990, also to replace organophosphate and carbamate insecticides. They function by binding nicotinic acetylcholine receptors in the central nervous system of invertebrates and are now the most widely used insecticides in the world (Mineau and Palmer 2013; Hallman et al. 2014). The function of neonicotinoids, and their ability to persist in the environment, make them highly toxic to invertebrate pollinators and a contributor to the decline of grassland birds (Mineau and Palmer 2013). They are also water soluble, meaning they have a high propensity for runoff and ground water infiltration (Hallman et al. 2014). Consequently, several neonicotinoid-based insecticides (e.g., imidacloprid and thiamethoxam) are on California's Groundwater Protection List (California Code of Regulations Title 3, Division 6, §6800).

Rodenticides

Rodenticides may also be used on agricultural sites to control known pests like mice (*Mus* spp.), roof rats (*Rattus rattus*), ground squirrels (*Spermophilus* spp.), and pocket gophers (*Thomomys* spp.; CDPR 2015). Anticoagulant rodenticides (ARs) function by inhibiting the synthesis of vitamin K in the liver, which delays coagulation and ultimately leads to hemorrhaging and death (Watt et al. 2005). There is a lag time between ingestion and death, meaning target species may ingest several toxic doses before they die (Herring et al. 2017). Effects of AR exposure can include acute poisoning, compromised immune systems, secondary poisoning through the consumption of exposed prey, and decreased ability to clot properly causing small injuries to bleed excessively (Gordon 1994). Anticoagulant rodenticides pose a threat to not only target and non-target wildlife, but also to children and pets—poison centers in the U.S. receive tens of thousands of reports of rodenticide exposure and ingestion annually (EPA 2011).

There are two types of ARs: first-generation and second-generation. Second-generation ARs were created after pest species began developing a resistance to first-generation ARs like warfarin (Hosea 2000). Second-generation ARs are more acutely toxic, more lipophilic (which increases their tissue accumulation and retention), and have longer half-lives (i.e., the time required for a concentration to decrease by half in a given organ like the liver; Hosea 2000; Erickson and Urban 2004). This means animals that ingest second-generation ARs can potentially carry the compound for years as compared to the shorter durations of first-generation ARs (CDPR 2018). Second-generation brodifacoum, bromadiolone, and difenacoum, for example, have hepatic half-lives of 113.5–350, 170–318, and 118 days, respectively, as compared to first-generation chlorophacinone, diphacinone, and warfarin that have half-lives of < 2, 3, and 26.2 days, respectively (CDPR 2018). The longer half-lives also mean that target pest species have the propensity to consume multiple doses of second-generation ARs prior to death, leading to the bioaccumulation (i.e., higher concentrations) of second-generation ARs in their organs, in turn posing a greater risk to the predators and scavengers that may consume them (Stone et al. 2003; Riley et al. 2007; Lima and Salmon 2010). As a result of documented exposure to non-target wildlife, second-generation ARs (i.e., brodifacoum, bromadiolone, difenacoum, and difethialone) have been labeled as California restricted materials (Table 2).

Acute rodenticides like bromethalin, cholecalciferol, strychnine, and zinc phosphide act more rapidly than ARs and are available to both professionals and homeowners. Acute rodenticides have varying modes of action, all of which differ from anticoagulants. Bromethalin, for example, decreases adenosine triphosphate synthesis, which leads to a buildup of cerebral spinal fluid, damage to the central nervous system, and lastly, paralysis, convulsions, and death (Van Lier and Cherry 1988). Strychnine, alternatively, blocks the uptake of glycine, an inhibitory neurotransmitter, which leads to increased motor neuron impulses, respiratory muscle spasms, and ultimately respiratory failure (Lawrence et al. 2009). Secondary poisoning from acute rodenticides is uncommon, but they do pose a significant hazard to anything that may consume them including nontarget wildlife (e.g., striped skunks- *Mephitis mephitis* and raccoons- *Procyon lotor*), people, livestock, and pets (van den Brink et al. 2018). Strychnine, for example, is used to control pocket gophers (*Thomomys* spp.) and is only legally applied underground. Improper use and deployment of strychnine, however, has resulted in non-target losses of birds and mammals alike (Littrell 1990). Numerous cases of bromethalin intoxication have been reported in urban wildlife, also likely from improper bait placement (McMillin et al 2016). Lastly, cholecalciferol has been found at illegal cannabis cultivation sites in northern California and was assumed to be a contributing factor in the death of a fisher (*Pekania pennanti*) given it had signs of hypercalcemia (Gabriel et al. 2015).

EFFECTS OF PESTICIDE EXPOSURE

Acute effects

Acute poisoning can follow direct exposure through ingestion, inhalation, or dermal contact, or secondary exposure through ingestion of contaminated prey (Berny 2007, CDPR 2018). Acute effects from pesticides have been documented in target and non-target species alike, ranging from insect pollinators and other arthropods to birds and mammals, and can ultimately result in decreased species diversity (Tables 1, 3; Clarke et al. 1986; Warner 1994).

Secondary exposure to anticoagulant rodenticides.—In California, secondary exposure of predators and scavengers to ARs appears to be widespread. Sixty-nine percent of wildlife collected by California's Department of Fish and Wildlife (CDFW) in 1994–1999 and 92% of mountain lions (*Puma concolor*) tested by CDFW in 2015–2016 tested positive for one or more ARs (Hosea 2000; Rudd et al. 2018). Additionally, 89% of raptors collected by a public health surveillance program in 2007, 73.5% of endangered San Joaquin kit foxes (*Vulpes macrotis mutica*) collected in Bakersfield from 1985-2009, and 89% of bobcats (*Lynx rufus*) necropsied by the National Park Service in 1997–2012 tested positive for one or more ARs (Lima and Salmon 2010; Cypher et al. 2014; Serieys et al. 2015). While rodenticides have

Table 3. Examples of acute effects of pesticide exposure on non-target wildlife species.

Taxa	Species	Response	Source
Birds	Bald eagle (*Haliaeetus leucocephalus*), red-tailed hawk (*Buteo jamaicensis*), great horned owl (*Bubo virginianus*)	Birds were tested for poisoning by organophosphate insecticides from 1984-85. Insecticides were the cause of death for 8 eagles, 2 hawks, and 1 owl.	Henny et al. 1987
	103 bird species in 12 orders	8,877 avian carcasses, where death was attributed to organophosphates and carbamates, were recovered between 1980 and 2000 in the United States.	Fleischli et al. 2004
	10 common raptor species	Raptors, none of which were suspected of having rodenticide exposure, were collected as part of a public health surveillance program in California. Of the 96 birds tested, 87% (n = 83) tested positive for one or more anticoagulant rodenticides.	Lima and Salmon 2010
	Barred owls (*Strix varia*), eastern screech owls (*Megascops asio*), great horned owl	Among the 161 owls that were admitted to a wildlife clinic in Massachusetts from 2006 – 2010, 138 (89%) tested positive for anticoagulant residues, predominantly brodifacoum.	Murray 2011
	Barn owl (*Tyto alba*)	119 owls from British Columbia were tested for rodenticide exposure. 29% of owls had toxicosis symptoms, males had higher concentrations of SGARs than females, and juveniles were more likely to show signs of toxicosis than adults.	Huang et al. 2016
	Red-tailed hawk	Among the 97 hawks that were caught, 8 tested positive for diphacinone. Tests were based on blood samples; the half-lives of anticoagulant rodenticides in the blood are much shorter than in the liver.	Abernathy et al. 2018
	Northern spotted owls (*Strix occidentalis caurina*), barred owls	7 out of 10 northern spotted owls and 34 out of 84 barred owls tested positive for anticoagulant rodenticide exposure. Barred owls were lethally removed as part of a management action and northern spotted owls were opportunistically collected.	Gabriel et al. 2018
Mammals	Coyote (*Canis latrans*)	During a 9-year study in southern California, anticoagulant rodenticide exposure was the second leading cause of mortality; 83% of coyotes tested were exposed.	Riley et al. 2003

Table 3. continued.

Taxa	Species	Response	Source
	San Joaquin kit fox (*Vulpes macrotis mutica*)	In foxes (n=68) collected in Bakersfield from 1985-2009, 73.5% of livers tested positive for AR exposure.	Cypher et al. 2014
	Bobcat (*Lynx rufus*), mountain lion (*Puma concolor*)	90% of bobcat livers (*n* = 39) and 100% of mountain lion livers (*n* = 4) tested positive for AR exposure, including brodifacoum, bromadialone, diphacinone, and/or difethialone.	Riley et al. 2007
	Bobcat	Livers from 172 bobcats were sampled for anticoagulant rodenticide exposure from 1997-2012 in southern California. 89% of the samples came back positive, including both sexes and age classes. Fetal transfer of rodenticides was also documented.	Serieys et al. 2015
	Mountain lion	Out of 64 lions tested by California Fish and Wildlife, 92% had detectable levels of anticoagulant rodenticides; 67% were exposed to first-generation rodenticides and 92% to second-generation.	Rudd et al. 2018
	American badger (*Taxidea taxus*), red fox (*Vulpes vulpes*)	2.2x more badgers and 6.4x more foxes were observed per km of road in a study area with low poisoning as compared to a study area with high poisoning. Poisoning was from strychnine- and chlorophacinone-treated oat baits.	Proulx and MacKenzie 2012
	Fisher (*Pekania pennanti*)	101 fishers were collected for necropsy in California during 2007-2014, of whom 86 were exposed to one or more anticoagulant rodenticides (= 1.73 AR compounds/indiv).	Gabriel et al. 2015
	Wood mice (*Apodemus sylvaticus*), bank voles (*Clethrionomys glareolus*), field voles (*Microtus agrestis*)	Coumatetralyl anticoagulant rodenticide bait, pre-mixed with a marker dye, was deployed for routine rat control. 48.6% of individuals in local populations of non-target small mammals ate the bait as indicated by the presence of the marker dye in their feces.	Brakes and Smith 2015

Table 3. continued.

Taxa	Species	Response	Source
Herpetofauna	Wood frogs (*Rana sylvatica*), leopard frogs (*R. pipiens*), green frogs (*R. clamitans*), bullfrogs (*R. catesbeiana*), American toads (*Bufo americanus*), gray tree frogs (*Hyla versicolor*)	In an experimental setting, exposure to carbaryl (six concentrations- 6.5, 3.2, 1.6, 0.3, 0.03, and 0.0 mg/L) became up to 8x more deadly in green frogs and up to 46x more lethal in bullfrogs when combined with predatory stress (i.e., adult red-spotted newts- *Notophthalmus viridescens*). Synergistic interactions were also documented in leopard frogs and toads but not in wood frogs or gray tree frogs.	Relyea 2003
	European common frog (*Rana temporaria*)	Frogs were exposed to three application rates (recommended max, 0.1x, and 10x label rate) of six pesticides including three fungicides (pyraclostrobin, captan, and spiroxamine), two herbicides (bromoxyniloctanoate and fenoxaprop-P-ethyl), and one insecticide (dimethoate). Acute mortality ranged from 20-100% (= 60%) within seven days when pesticides were applied at the label rate.	Brühl et al. 2013
Multi-taxa	Mammals and birds	From 1971 – 1997 necropsies were done for wildlife submitted to a rehabilitation center in New York. Death from hemorrhage associated with anticoagulant rodenticides was documented in 51 cases; species ranged from owls to deer.	Stone et al. 1999
	Mammals and birds	Tissues from 74 dead animals were collected from 1994 – 1999 in California. 30 out of 43 mammals and 21 out of 31 birds examined ($n = 21$ species total) had rodenticide residues. Coyotes, bobcats, golden eagles (*Aquila chrysaetos*), and barn owls were among the species most frequently exposed.	Hosea 2000

been documented to negatively impact a wide array of non-target wildlife (Table 1), their impacts are particularly prevalent in the very species that help control rodent populations naturally like scavenging and predatory raptors (e.g., barn owl - *Tyto alba* and red-tailed hawk - *Buteo jamaicensis*) and mammals (e.g., bobcats and coyotes- *Canis latrans*; Gabriel et al. 2015; Elliott et al. 2016).

Secondary exposure to ARs has also been suggested as a contributing factor in the decline of threatened and endangered species (Gabriel et al. 2015, 2018). Northern spotted owls (*Strix occidentalis caurina*), for example, are a federal- and state-listed endangered species. One of their primary prey sources in northern California are dusky-footed woodrats (*Neotoma fuscipes*), which are also perceived as a threat to cannabis cultivation given that they forage on cannabis plants and use them as nesting material (Franklin et al. 2018). If ARs are used to control woodrats, particularly second-generation ARs at illegal cultivation sites, then the owls are at risk of secondary rodenticide exposure. In fact, Gabriel et al. (2018) found that 7 out of 10 northern spotted owls and 34 out of 84 barred owls (*S. varia*), an ecologically similar species, tested positive for AR exposure in northern California where thousands of illegal cannabis cultivation sites have been documented on private and public lands.

Toxicosis from anticoagulant rodenticides.—Cases of lethal poisoning from ARs in non-target wildlife are much rarer than secondary exposure. For a mortality to be attributed to AR exposure, the animal must have acute clinically significant signs of toxicant exposure including detectable levels of AR(s) in the liver, coagulopathy, and hemorrhaging that cannot be attributed to any other causes (Gabriel et al. 2015). One of the most notable documentations of a non-target wildlife species dying of toxicosis is fishers in California (Gabriel et al. 2012, 2015). Gabriel et al. (2015) found that 13 fishers had died of toxicosis, 11 from ARs specifically, and that the source of the rodenticides was likely illegal cannabis cultivation sites.

Acute effects of insecticides.—Acute effects from insecticides have also been documented in numerous non-target wildlife species (Relyea 2003; Fleischli et al. 2004). Insecticides, similar to ARs, tend to reduce the population sizes of predators and parasites that help control plant-feeding arthropods naturally like grassland birds (Pimentel 2005; Mineau and Palmer 2013). In addition, exposure to insecticides can contribute to the decline of threatened and endangered species. A study by Davidson and Knapp (2007) found that use of anticholinesterase insecticides upwind of sampled sites had a significant, negative influence on the probability that mountain yellow-legged frogs (*Rana muscosa*), a threatened species, would be present. They also found that the landscape-scale effect of anticholinesterase insecticides was stronger than that of fish, the primary variable (i.e., the introduction of nonnative fish to historically fishless areas) that has been attributed to the decline of yellow-legged frogs (Davidson and Knapp 2007). Amphibian species in general may be especially prone to pesticide exposure given their skin is highly permeable and the life cycle of some species encompasses aquatic and terrestrial phases, meaning they may be exposed to pesticides in two environments (Brühl et al. 2013).

Sublethal effects

Equally concerning to the acute poisoning of wildlife populations are the physiological, phenological, and behavioral impacts associated with sublethal exposure to pesticides (Table 4; Baldwin et al. 2009; Fraser et al. 2018). These impacts are cryptic in that they may

Table 4. Examples of sublethal effects of pesticide exposure on non-target wildlife species.

Taxa	Species	Response	Source
Birds	American kestrel (*Falco sparverius*)	Kestrels experienced hypothermia when they were given a single oral dose of the organophosphate methyl parathion (2.25 mg/kg wt) and were exposed to 10 hours of cold (temperature was normally within their thermoneutral zone).	Rattner and Franson 1983
	European starlings (*Sturnus vulgaris*)	Males decreased their singling and displaying by 50% following an oral dose of the organophosphate dicrotophos (2.5 mg/kg wt), which could reduce their reproductive success.	Grue and Shipley 1984
	European starlings	When given a single dose of the organophosphate dicrotophos, adults lost an average of 14% of their initial body weight within 24-hr of the dose.	Grue and Shipley 1984
	Northern bobwhites (*Colinus virginianus*)	After being fed a diet with varying concentrations of the organophosphate methamidophos, birds consumed less food and in turn, produced fewer eggs and had smaller clutch sizes.	Stromborg 1986
	Mallards (*Anas platyrhynchos*)	Ducklings exposed to low levels of carbofuran (0.22 mg/kg wt) experienced hypothermia and increased mortality at temperatures as high as 50° F.	Martin and Solomon 1991
	Mallards	Including methyl parathion (400 ppm) in the diet of mallards for 8 days resulted in a 50% reduction in egg production. The ducks also consumed only 16% of that of controls.	Bennett et al. 1991
	House sparrow (*Passer domesticus*)	After being exposed to a single sublethal dose of the organophosphate fenthion, sparrows were 16x more likely to be predated than controls within the same flock.	Hunt et al. 1992
	White-throated sparrow (*Zonotrichia albicollis*)	When given an oral dose of the organophosphate fenitrothion, birds fledged at lower body weights than controls.	Grue et al. 1997
	Insectivorous birds	The average intrinsic rate of increase in local bird populations was negatively associated with the concentration of imidacloprid, a neonicotinoid insecticide, likely because the birds ($n = 15$ species) feed their young almost exclusively with invertebrates.	Hallman et al. 2014
Mammals	Bobcats (*Lynx rufus*)	All bobcats with advanced mange ($n = 19$) had anticoagulant rodenticide compounds in their liver. Further, rodenticide levels were higher in bobcats that died of mange than those that died of other causes.	Riley et al. 2007
	Bobcat	Samples were collected from 124 bobcats presumed to be healthy from 2007 – 2012. Anticoagulant rodenticide exposure was positively associated with B-cell counts (i.e., the immune component that targets invading pathogens) and negatively associated with creatine (i.e., an indicator of kidney function).	Serieys et al. 2018

Table 4. continued.

Taxa	Species	Response	Source
	Common vole (*Microtus arvalis*)	Voles trapped in alfalfa fields treated with the anticoagulant rodenticide chlorophacinone had higher prevalence of the bacteria that causes tularemia, a zoonotic disease, than voles trapped in control fields.	Vidal et al. 2009
Herpetofauna	Wood frogs (*Rana sylvatica*)	Frogs were more susceptible to trematode infections in ponds with greater exposure to agricultural runoff, likely because of increased stress due to exposure to organochlorine pesticides and organophosphorus compounds (e.g., Altrazine and Malathion). This, in turn, led to a higher risk of limb deformities.	Kiesecker 2002
	African clawed frogs (*Xenopus laevis*)	Larvae were exposed to atrazine (0.01 – 200 ppb), a commonly used herbicide. Atrazine induced hermaphroditism, demasculinized the larynxes of males, and resulted in an up to 10-fold decrease in male's testosterone levels.	Hayes et al. 2002
	Leopard frogs (*R. pipiens*)	Frogs were exposed to a mixture of atrazine, metribuzin, aldicarb, endosulfane, lindane, and dieldrin for 21 days and then challenged with a parasitic nematode (*Rhabdias ranae*). Frogs exposed to pesticides had reduced T-cell proliferation (i.e., cells that defend against extracellular parasites) and more nematodes in their lungs than control animals.	Christin et al. 2003
	Green frogs (*R. clamitans*)	The net effect of exposure to glyphosate (3700 μg/L) and atrazine (201 μg/L), two commonly used herbicides, and to malathion (9.6 μg/L) and carbaryl (33.5 μg/L), two broad-based insecticides, was to increase frog susceptibility to trematode (*Echinostoma trivolvis*) infections.	Rohr et al. 2008
	Leopard frogs	Outdoor mesocosms that contained zooplankton, phytoplankton, periphyton, and larval amphibians were exposed to low concentrations of malathion, an insecticide. The malathion caused a decrease in zooplankton and subsequently an increase in phytoplankton, decrease in periphyton, and decrease in the growth and development of leopard frogs, which rely on periphyton as their food source.	Relyea and Diecks 2008
	Wood frogs, leopard frogs, American toads (*Bufo americanus*)	Outdoor mesocosms that included leaf litter, algae, zooplankton, and three species of tadpoles were exposed to Roundup herbicide and/or predator cues (i.e., newts or larval dragonflies). Roundup resulted in morphological changes (i.e., increase in tail depth) like the adaptive changes induced by dragonfly (i.e., predator) cues.	Relyea 2012

reduce individual fitness and population persistence without the obvious signs of pesticide exposure (Fraser et al. 2018).

Immunological effects.—One sublethal impact may be to an individual's immune system (Li and Kawada 2006; Zabroskii et al. 2012; Serieys et al. 2018). Insecticides like endosulfan and malathion have been found to compromise individuals' immune systems, leaving the animal more susceptible to parasitic infections and pathogens (Christin et al. 2003; Rohr et al. 2008). Leopard frogs (*Rana pipiens*) that were exposed to a mixture of four insecticides and two herbicides, for example, had fewer T-lymphocytes (i.e., one of the immune system cells that fight off extracellular parasites) and consequently, were at greater risk to parasitic nematodes (Christin et al. 2003). One study suggested that bobcats exposed to anticoagulant rodenticides, alternatively, experienced changes in their immune responses to allergens, tumors, viral infections, and novel pathogens (Serieys et al. 2018). Further, given immune responses are energetically costly, regular immune stimulation from anticoagulant rodenticide or insecticide exposure may lead to an overall decrease in fitness (Serieys et al. 2018).

Reproductive effects.—Pesticides may also adversely affect reproduction. Exposure to certain types of insecticides (e.g., organophosphates and neonicotinoids) has been documented to cause embryotoxicity (i.e., the transfer of residual pesticides from the parent to young), chemical castration, and a reduction in courtship displays, all of which can lead to reproductive failure (Grue and Shipley 1984; Grue et al. 1997; Pimentel 2005; Mineau and Palmer 2013). Additionally, mammals and birds have been shown to have reduced litter and clutch sizes, perhaps because insecticide and rodenticide exposure can decrease an animal's appetite (Bennett et al. 1991; Erickson and Urban 2004).

Thermoregulatory effects.—Sublethal exposure to pesticides like anticholinesterase insecticides can also impair thermoregulatory abilities, which can lead to death (Gordon 1994; Grue et al. 1997). Mallard ducklings (*Anas platyrhynchos*), for example, experienced hypothermia after being exposed to a low dose of carbofuran, with some ducklings dying at temperatures as high as 10° C, a temperature that would typically be within the species' thermoneutral zone (Martin and Solomon 1991). This suggests that pesticide exposure may pose an even greater risk to wildlife when weather conditions fall outside of the species' thermoneutral zone (Rattner and Franson 1983; Martin and Solomon 1991).

Morphological effects.—Altered morphology following exposure to certain types of insecticides has primarily been documented in amphibians. In a meta-analysis of experimental studies aimed at measuring the effects of chemical pollutants on amphibians, researchers found that the overall effect size of pollutants was a 535% increase in the frequency of abnormalities (e.g., limb deformities) as well as a 14.3% decrease in survival and 7.5% decrease in mass (Egea-Serrano et al. 2012). Frogs have also been documented to have inhibited growth of the larynx and to develop morphological phenotypes that are poorly suited for their environment (Kiesecker 2002; Relyea and Diecks 2008; Relyea 2012).

Behavioral effects.—Lastly, pesticides may alter the behavior, composition, and abundance of both predators and prey (e.g., insects and small mammals). Following pesticide exposure, arthropods exhibit altered search and attack behaviors, mammals have decreased coordination, motor skills, and response times, and fishes can develop swimming abnormalities, all of which make the respective individual more susceptible to predation (Pimentel 2005; Wolansky and Harrill 2008; Sanchez-Barbudo et al. 2012; Renick et al. 2015). Animals that are secondarily exposed to pesticides may also be at greater risk to predation if they too experience responses like reduced mobility and response times (Serieys et al. 2015). If pesticides reduce the availability of food resources in a landscape, alternatively, then there can be cascading impacts throughout the food chain. In areas treated with insecticides, for example, insect prey populations like mosquitos and beetles have been shown to decline, which in turn has resulted in declines in the survival and abundance of insectivorous bird populations (Warner 1994; Hallmann et al. 2014).

REGULATIONS PERTAINING TO PESTICIDE USE ON PERMITTED CANNABIS CULTIVATION SITES IN CALIFORNIA

Commercial, adult-use cannabis cultivation was legalized in California in 2018. With legalization came a multitude of regulations pertaining to pesticide use. California state regulations restrict the use of pesticides in or around permitted cannabis cultivation if they are a) California restricted materials, b) on the ground water protection list, or c) not registered for a food use in California. California restricted materials (Table 2; California Code of Regulations Title 3, Division 6, §6400) are pesticides deemed to have a high potential to harm public health and the environment. They can only be purchased and used by, or under the supervision of, a certified applicator who has a permit issued by the County Agricultural Commissioner (CDPR 2014). The ground water protection list, alternatively, identifies pesticides that have the potential to pollute ground water (California Code of Regulations Title 3, Division 6, §6800), and restricts their use either statewide or in specified vulnerable areas (restrictions are pesticide-specific). Similar to other agricultural crops, cannabis cultivators must also: (1) comply with all pesticide label directions; (2) store chemicals in a secure building or shed to prevent access by wildlife; (3) contain any chemical leaks and immediately clean up any spills; (4) apply the minimum amount of product necessary to control the target pest; (5) prevent offsite drift; (6) not apply pesticides when pollinators are present; (7) not allow drift to flowering plants attractive to pollinators; (8) not spray directly onto surface water or allow pesticide product to drift to surface water and spray only when wind is blowing away from surface water bodies; (9) not apply pesticides when they may reach surface water or groundwater; and (10) only use properly labeled pesticides and consult with California Department of Pesticide Regulation (CDPR) if no label is available (CDFA 2019).

Regulations pertaining to pesticide use in cannabis cultivation are even more stringent when you incorporate federal regulations, or in this case the lack thereof. The United States Environmental Protection Agency (EPA) does not recognize cannabis as being a part of an existing crop group given it is illegal under federal law. This means there are no U.S. EPA-approved pesticide products for use on cannabis or U.S. EPA residue tolerance requirements (i.e., the amount of pesticide residue allowed to remain in or on each treated crop). Consequently, the only pesticides that can be legally applied to cannabis under California state law are pesticides with active ingredients that are exempt from residue tolerance requirements and either exempt from registration requirements or registered for a use broad enough to encompass cannabis (CDPR 2017). Most of these exempt pesticides are biorational like citronella or food-grade essential oils like cinnamon, garlic, and rosemary oils (CDPR 2017).

To monitor pesticide use on permitted cannabis cultivation sites, California's Bureau of Cannabis Control text of regulations (California Code of Regulations Title 16, Division 42) requires that cultivators have a pest management plan that includes "product names and active ingredient(s) of all pesticides to be applied to cannabis during any stage of plant growth" and "integrated pest management protocols including chemical, biological, and cultural methods the applicant anticipates using to control or prevent the introduction of pests on the cultivation site." The Bureau of Cannabis Control also requires cannabis cultivators to submit 0.5 g of every cannabis batch to be tested for Category I (i.e., not registered for food use in California) and Category II Residual Pesticides (Tables 5, 6). If the sample exceeds any of the threshold values, then the batch from which the sample was taken will not be released for retail sale.

Table 5. Category I Residual Pesticides (California Code of Regulations Title 16, Division 42, §5719.1) that are not registered for food use in California (i.e., they cannot be used in or around cannabis cultivation sites), the pesticide type, and the substance group (IUPAC 2019). When a cannabis batch is tested for residual pesticides, the limit of quantitation is 0.10 µg/g or lower for all Category I Residual Pesticides.

Residual pesticide	Pesticide type	Substance group
Aldicarb[a,b]	Insecticide / Acaricide / Nematicide	Carbamate
Brodifacoum[a]	Rodenticide	Hydrocoumarin
Bromadiolone[a]	Rodenticide	Coumarin anticoagulant
Carbofuran[a]	Insecticide / Acaricide / Nematicide	Carbamate
Chlordane	Insecticide	Organophosphate
Chlorfenapyr	Insecticide / Acaricide / Miticide	Pyrrole
Chlorpyrifos[a*]	Insecticide	Organophosphate
Coumaphos	Ectoparasiticide	Organothiophosphate
Daminozide	Plant growth regulator	Unclassified
DDVP (Dichlorvos)	Insecticide / Acaricide	Organophosphate
Difenacoum[a]	Rodenticide	Hydroxycoumarin
Difethialone[a]	Rodenticide	Coumarin anticoagulant
Dimethoate[b]	Insecticide / Acaricide	Organophosphate
Ethoprop(hos)[a*,b]	Insecticide / Nematicide	Organophosphate
Etofenprox	Insecticide	Pyrethroid
Fenoxycarb	Insecticide	Carbamate
Fipronil	Insecticide	Phenylpzrazole
Imazalil	Fungicide	Imidazole
Methiocarb[b]	Insecticide / bird repellant	Carbamate
Methyl parathion	Insecticide / Acaricide	Organophosphate
Mevinphos[a]	Insecticide / Acaricide	Organophosphate
Paclobutrazol	Fungicide	Triazole
Propoxur	Insecticide / Acaricide	Carbamate
Spiroxamine	Fungicide	Morpholine
Thiacloprid	Insecticide	Neonicotinoid

[a] California restricted material

[a*] California restricted material with exceptions

[b] On the Groundwater Protection List

Table 6. Category II Residual Pesticides (California Code of Regulations Title 16, Division 42, §5719.1), the threshold value that provides the criterion for determining whether a cannabis sample passes or fails an analytical test by the Bureau of Cannabis Control (i.e., Action level), the pesticide type, and the substance group (IUPAC 2019).

Residual pesticide	Action level (µg/g)		Pesticide type	Substance group
	Inhalable cannabis goods	Other cannabis goods		
Abamectin	0.1	0.3	Insecticide	unclassified
Acephate[b]	0.1	5	Insecticide	Organophosphate
Acequinocyl	0.1	4	Acaricide	unclassified
Acetamiprid	0.1	5	Insecticide	Neonicotinoid
Azoxystrobin[b]	0.1	40	Fungicide	Strobilurin
Bifenazate	0.1	5	Insecticide/ Acaricide	Hydrazine carboxylate
Bifenthrin	3	0.5	Insecticide/ Acaricide	Pyrethroid
Boscalid[b]	0.1	10	Fungicide	Carboxamide
Captan	0.7	5	Fungicide / Bactericide	Phthalimide
Carbaryl[a*,b]	0.5	0.5	Insecticide	Carbamate
Chlorantraniliprole[b]	10	40	Insecticide	Anthranilic diamide
Clofentezine	0.1	0.5	Acaricide	Tetrazine
Cyfluthrin[a]	2	1	Insecticide	Pyrethroid
Cypermethrin	1	1	Insecticide	Pyrethroid
Diazinon[b]	0.1	0.2	Insecticide / Acaricide	Organophosphate
Dimethomorph[b]	2	20	Fungicide	Morpholine
Etoxazole	0.1	1.5	Acaricide	Diphenyl oxazoline
Fenhexamid	0.1	10	Fungicide	Hydroxyanilide
Fenpyroximate	0.1	2	Insecticide / Acaricide	Pyrazolium
Flonicamid	0.1	2	Insecticide / Aphicide	Pyridine
Fludioxonil[b]	0.1	30	Fungicide	Phenylpyrrole
Hexythiazox	0.1	2	Acaricide	Carboxamide
Imidacloprid[b]	5	3	Insecticide	Neoicotinoid
Kresoxim-methyl	0.1	1	Fungicide / Bactericide	Strobilurin
Malathion[b]	0.5	5	Insecticide / Acaricide	Organophosphate
Metalaxyl[b]	2	15	Fungicide	Phenylamide
Methomyl[a*,b]	1	0.1	Insecticide / Acaricide	Carbamate
Myclobutanil[b]	0.1	9	Fungicide	Triazole
Naled[a]	0.1	0.5	Insecticide / Acaricide	Organophosphate
Oxamyl	0.5	0.2	Insecticide / Acaricide	Carbamate
Permethrin	0.5	20	Insecticide	Pyrethroid
Phosmet	0.1	0.2	Insecticide / Acaricide	Organophosphate

Table 6. continued.

Residual pesticide	Action level (µg/g)		Pesticide type	Substance group
	Inhalable cannabis goods	Other cannabis goods		
Piperonyl butoxide	3	8	Other	Cyclic aromatic
Prallethrin	0.1	0.4	Insecticide	Pyrethroid
Propiconazole[b]	0.1	20	Fungicide	Triazole
Pyrethrins	0.5	1	Insecticide	unclassified
Pyridaben	0.1	3	Insecticide / Acaricide	Pyridazinone
Spinetoram	0.1	3	Insecticide	Spinosym
Spinosad	0.1	3	Insecticide	Natural substance
Spiromesifen	0.1	12	Insecticide	Tetronic acid
Spirotetramat	0.1	13	Insecticide	Tetronic acid
Tebuconazole[b]	0.1	2	Fungicide	Triazole
Thiamethoxam[b]	5	4.5	Insecticide	Neonicotinoid
Trifloxystrobin	0.1	30	Fungicide	Strobilurin

[a] California restricted material

[a*] California restricted material with exceptions

[b] On California's Groundwater Protection List

The legalization of commercial, adult-use cannabis cultivation came with a bountiful number of pesticide regulations. These regulations aim to help minimize potential environmental impacts of permitted cannabis cultivation and are one of the many ways in which legal cultivation is delineated from illegal cultivation. At legal cultivation sites, for example, the toxic pesticide products that tend to result in acute and sublethal effects in non-target wildlife species cannot be legally applied. At illegal cultivation sites, alternatively, the use of these toxic pesticides and numerous ensuing environmental impacts have been well documented (Gabriel et al. 2012, 2015, 2018). Given legal cannabis cultivation is still in its infancy in California, however, there are many knowledge gaps. We encourage studies focused on the types and quantities of pesticides being used at permitted cannabis cultivation sites and assessments of whether cultivators are fully abiding to regulations. We also encourage studies aimed at improving our understanding of how pesticide use in cannabis cultivation relates to other agricultural industries, and if there are any pesticide-related impacts unique to cannabis given it tends to be grown in rural and forested areas (Butsic et al. 2018). Lastly, we urge studies comparing pesticide use at legal vs. illegal cannabis cultivation sites and documentation of any subsequent environmental impacts as this information would likely highlight some of the benefits of legalization.

Author contributions

Conceived and designed the study: LNR, ADB
Collected the data: LNR, ADB
Performed the analysis of the data: LNR, SM, ADB
Authored the manuscript: LNR, SM, ADB, EC
Provided critical revision of the manuscript: SM, ADB, EC

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An example of an indoor cannabis cultivation in San Diego County enrolled in the State Water Resources Control Board's Cannabis Cultivation Program. Indoor facilities such as this are issued a waiver of waste discharge requirements. The photo shows approximately 2,100 cannabis plants. Photo credit: Kyle Castanon, Palomar Craft Cannabis

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Two years after legalization: implementing the Cannabis Cultivation Policy in southern coastal California

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During the first two years of legal recreational cannabis cultivation in California, the State Water Resources Control Board's Cannabis Cultivation Program enrolled 4,391 cultivators into the Cannabis General Order. The South Coast Regional Cannabis Unit, covering approximately 28,500 km^2 in southern coastal California, account for 519 of those cultivators, 516 of which are indoor cultivations. The observed distribution of commercial recreational cannabis cultivation reflects local government restrictions and, combined with the urban nature of the areas where cultivation is permitted in southern California, results in a majority of cultivation being indoor. Of the active enrollees in the South Coast region with indoor cultivation, 94% of the cultivations are discharging their industrial wastewater to a publicly owned treatment works via a sewer connection. The remaining enrollees in the South Coast region with indoor cultivation haul their industrial wastewater to a permitted wastewater treatment facility. These discharges pose a low threat to water quality, provided that they are compliant with the sewer agency's requirements and/or wastewater treatment facility's requirements. As a result, all but the three outdoor cultivators enrolled with the State Water Resources Control Board in the South Coast region were issued waivers of waste discharge requirements as of 1 January 2020.

Enforcement of illicit cultivation for the purpose of the protection of water quality and habitat also plays an important role in the South Coast Regional Cannabis Unit's Cannabis Cultivation Program. The South Coast Regional Cannabis Unit inspected 143 illicit cultivation sites as of 1 January 2020. Observations from inspections of illicit sites identified threats to Clean Water Act 303(d) waterbodies and Regional Water Quality Control Board priority water systems, and observed illegal and/or unauthorized pesticide use, evidence of and potential for sediment mobilization, and ubiquitous and substantial litter. These illicit cultivation sites pose a threat to water quality, aquatic and riparian habitats, and fish and wildlife, though the effects have not yet been quantified.

Key Words: California Regional Water Quality Control Boards, cannabis, cultivation, environmental impacts, management, water quality

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California has a long history of cannabis cultivation with records of cultivation dating back to 1795 when hemp was introduced to Mission San Jose (Bowman 1943, Clarke and Merlin 2013). Hemp plantations supported by the Spanish government in colonies within California were established in Santa Barbara, San Jose, Los Angeles and San Francisco in the early 1800's (Clarke and Merlin 2013).

Enacted by the 91st United States Congress, the Comprehensive Drug Abuse Prevention and Control Act of 1970 (21 United States Code (USC) Chapter 13 §812) established cannabis ("marihuana") as a Schedule I controlled substance[1]. California was the first state to legalize non-hemp cannabis cultivation. Legalization of medical cannabis cultivation in California began in 1996 with the passage of Proposition 215 the Compassionate Use Act of 1996 (Health and Safety Code (HSC) §11362.5; HSC §11358). The Medical Marijuana Regulation and Safety Act of 2016, and later the Medicinal and Adult-Use Cannabis Regulation and Safety Act (MAUCRSA) by initiative Proposition 64, established the general framework for regulation of cannabis in California, including its cultivation (Business and Professions Code (BPC) §26000). MAUCRSA authorized the California Department of Food and Agriculture (CDFA) to administer provisions related to cultivation in the state (BPC §26012). CDFA, while the ultimate licensing board for cultivation, is required to include conditions requested by the California State Water Resources Control Board (SWRCB) and California Department of Fish and Wildlife (CDFW) (BPC §26060.1(b)(1); Water Code §13149). While cultivation for personal use is legal statewide under Proposition 64, restrictions to cultivation within HSC §11362.2 are ultimately in the hands of individual local governments, counties, cities, and special districts.

Extensive literature has been produced documenting cannabis cultivation in northern California (Levy 2014, Carah et al. 2015, Butsic and Brenner 2016, Butsic et al. 2018, Wilson et al. 2019, Levy 2020), especially in Humboldt, Mendocino, and Trinity counties known collectively as the Emerald Triangle, however significantly less has been prepared documenting cannabis cultivation in southern California. What follows is a compilation of data describing cannabis cultivation in southern California based on the initial two years of records with the California Regional Water Quality Control Board's South Coast Regional Cannabis Unit (SCRCU). The data set includes 662 cannabis cultivation sites, both legally and illegally operating in southern California.

Statewide Regulatory Framework

In the decades following the passage of Proposition 215 the Compassionate Use Act, cannabis cultivation in northern California rapidly expanded, resulting in waste discharges to the environment, loss of instream flows, and the diversion of springs and streams (North Coast Regional Water Quality Control Board 2015, Central Valley Regional Water Quality Control Board 2015). To mitigate these environmental impacts, the North Coast Regional Water Quality Control Board (RWQCB) and the Central Valley RWQCB began regulating medicinal cannabis cultivation in 2015 under SWRCB Orders R1-2015-0023 and R5-2015-0113, respectively (North Coast Regional Water Quality Control Board 2015, Central Valley Regional Water Quality Control Board 2015). These orders laid the framework for the statewide cannabis cultivation policy after the passage of MAUCRSA, when recreational cannabis cultivation became legal in the state.

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[1] Does not include hemp as defined in 7 USC §1639o

The SWRCB adopted Resolution Number 2017-0063, the Cannabis Cultivation Policy – Principles and Guidelines for Cannabis Cultivation (Cannabis Policy), on 17 October 2017 which became effective on 18 December 2017 (State Water Resources Control Board 2017a). The Cannabis Policy was subsequently updated by Resolution Number 2019-0007 (State Water Resources Control Board 2019a):

"The purpose of [the] Cannabis Policy is to ensure that the diversion of water and discharge of waste associated with cannabis cultivation does not have a negative impact on water quality, aquatic habitat, riparian habitat, wetlands, and springs." (State Water Resources Control Board 2019a)

The Cannabis Policy achieves this purpose by establishing requirements for cannabis cultivation that address the following 12 treatment or control categories (State Water Resources Control Board 2019a):

- riparian and wetland protection and management;
- water diversion, storage, and use;
- irrigation runoff;
- land development and maintenance, erosion control, and drainage features;
- soil disposal;
- stream crossing installation and maintenance;
- fertilizer and soil use and storage;
- pesticide and herbicide application and storage;
- petroleum products and other chemical use and storage;
- cultivation-related waste disposal;
- refuse and human waste disposal; and
- winterization

The Cannabis Policy applies to commercial recreational cannabis and commercial medical cannabis, and exempts personal use recreational or personal use medical cannabis cultivation, as long as the cultivation complies with applicable conditions including riparian setback requirements and disturbed area limitations (State Water Resources Control Board 2019a). The requirements of the Cannabis Policy are implemented through SWRCB Order WQ 2017-0023-DWQ (State Water Resources Control Board 2017b), General Waste Discharge Requirements and Waiver of Waste Discharge Requirements for Discharges of Waste Associated with Cannabis Cultivation Activities (Cannabis General Order), which was updated by SWRCB Order WQ 2019-0001-DWQ (State Water Resources Control Board 2019b). The Cannabis General Order established a tiered approach based on the potential threat to water quality of cannabis cultivation and related activities. Enrollees in the Cannabis General Order are issued general waste discharge requirements (WDRs) that address issues including the mobilization of sediment, the use of fertilizers, and storm water runoff. The tiers are based on the size of the disturbed area, as defined in the Cannabis Policy, the slope of the disturbed area, and the proximity to a water body. The Cannabis General Order also established two conditional exemptions that are issued conditional waivers of WDRs, an indoor commercial cultivation exemption and an outdoor commercial cultivation exemption if the disturbed area for cultivation is less than 2,000 ft^2 (~185 m^2). Indoor cultivation is defined as activities that occur within a structure with a permanent roof, a permanent relatively impermeable floor, such as concrete or asphalt paved, and either discharges all industrial wastewaters to a permitted wastewater treatment facility that accepts cannabis cultivation wastewater, or collects all industrial wastewater in a storage container and hauls the wastewater offsite to a permitted wastewater treatment facility (State Water Resources Control Board 2019a).

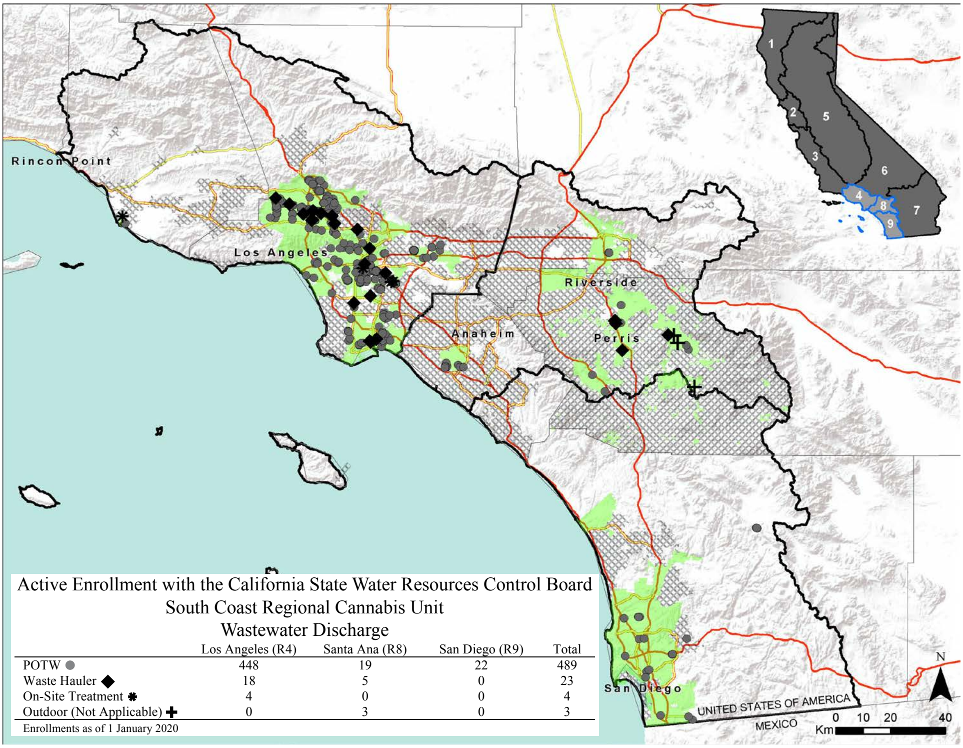
South Coast Regional Cannabis Unit Regional Setting

The SWRCB consists of nine RWQCBs based on watershed boundaries, shown on Figure 1. To account for California's size and geographical variations, which result in highly variable climate and drainage patterns, the Cannabis Policy defines 14 Cannabis Cultivation Policy regions and identifies nine priority regions that support anadromous salmonids where water flow variability and water quality have the potential to inhibit natural migration, spawning, and rearing (State Water Resources Control Board 2019a). The 14 Cannabis Cultivation Policy regions are divided among six cannabis cultivation regulatory units based on geographic location. The SCRCU's Policy region includes one of the nine priority Cannabis Cultivation Policy regions and is based out of the Santa Ana RWQCB in Riverside, California. The SCRCU implements the SWRCB's Cannabis Regulatory program for the Santa Ana, Los Angeles, and San Diego RWQCBs. The area regulated by the SCRCU and the subject of this study spans approximately 28,500 km^2 and seven counties in southern coastal California, from Rincon Point, Ventura County in the north to the United States-Mexico border in the south, as shown on Figure 1.

The South Coast region lies within the Transverse and Peninsular Ranges geomorphic provinces. Plate motions between the Pacific Plate and North American Plate shape the region's dramatic landscape, where elevations range from sea level to 3,500 m above sea level. Major mountain ranges bound the South Coast region on the north and east and drain water and sediment into the multiple inland and coastal basins. The South Coast region aligns with the Southern California Coastal (Hydrologic Unit Code [HUC] 1807) Subregion and encompasses the Ventura-San Gabriel Coastal (HUC 180701), Santa Ana (HUC 1800702), and Laguna-San Diego Coastal (HUC 180703) Basins, 17 Subbasins (HUC8), 82 Watersheds (HUC10), 308 Subwatersheds (HUC12) (United States Department of Agriculture 2019), and 75 groundwater basins (DWR 2018). Broadly speaking, surface streams drain south-southwestward to the Pacific Ocean or to endorheic inland basins. Average annual precipitation within the South Coast region ranges from less than 10 inches in the inland valleys and near the border with Mexico to 40 inches in parts of the Transverse Ranges (United States Environmental Protection Agency 2019). The majority of surface waters are intermittent or ephemeral streams and are dry for most of the year.

METHODS

This data set is a compilation of records with the SWRCB and SCRCU from December 2017 to 1 January 2020. The SCRCU conducted inspections of cannabis cultivators throughout the South Coast region. This data set includes observations made by SCRCU staff from 24 compliance inspections in 2019 at cannabis cultivators actively enrolled within the SWRCB's Cannabis Policy and 143 inspections of illicit cannabis cultivation sites dating from 29 March 2018 through 1 January 2020. Requirements of the Cannabis Policy and Cannabis General Order are enforceable under the Porter-Cologne Water Quality Control Act (Water Code §13000) and site inspection authority of enrolled cannabis cultivators is provided in the Cannabis Policy. Site inspections of illicit cultivations were conducted



Active Enrollment with the California State Water Resources Control Board
South Coast Regional Cannabis Unit
Wastewater Discharge

	Los Angeles (R4)	Santa Ana (R8)	San Diego (R9)	Total
POTW ●	448	19	22	489
Waste Hauler ◆	18	5	0	23
On-Site Treatment ✱	4	0	0	4
Outdoor (Not Applicable) ✚	0	3	0	3

Enrollments as of 1 January 2020

Figure 1. As of 1 January 2020, 519 cultivators are enrolled with the SCRCU into the SWRCB's Cannabis Policy. The distribution of cultivators is shown throughout the region as a function of their mechanism of wastewater discharge; discharge to the sewer (circles), transport off site via waste hauler (diamonds), and onsite treatment (asterisks) are permitted mechanisms for indoor cultivators to dispose of wastewater. Outdoor cultivators (crosses) are issued waste discharge requirements (WDRs) or a conditional waiver of requirements depending on their potential risk to the environment. The distribution of cultivators throughout the South Coast region is geographically limited by the few local municipalities that allow cultivation (green shaded areas). Throughout most of the South Coast region, commercial cannabis cultivation is prohibited (hatched areas). The areas without shading are either federal lands or do not have ordinances governing cannabis cultivation as of 1 January 2020. The SWRCB's nine regional waterboards are shown in the inset, 1 – North Coast, 2 – San Francisco Bay, 3 – Central Coast, 4 – Los Angeles, 5 – Central Valley, 6 – Lahontan, 7 – Colorado River, 8 – Santa Ana, and 9 – San Diego.

in coordination with local law enforcement agencies under search warrants and typically included representatives from the CDFW and county code enforcement divisions.

Hereafter, a site refers to a property where a single inspection was performed by SCRCU staff. A site, as defined, may contain more than one address, assessor parcel number, and/or landowner. The observations discussed are only representative of what was noted on the day of each inspection. Sites are dynamic and ever changing. It should be noted that the data set represents only sites where the SCRCU were invited to participate in inspections by enrolled dischargers for permitted facilities or by local law enforcement for illicit cultivation facilities.

RESULTS

Enrollees in the South Coast Cannabis Cultivation Policy Region

As of 1 January 2020, the SCRCU has 519 active enrollees in the SWRCB's Cannabis Policy. Of the permits issued, 516 (99%) are indoor cannabis cultivations. The remaining three enrollees are either Tier 1 – Low risk outdoor cultivations (disturbed area between 2,000 and 43,560 ft^2; ~185 and ~4,046 m^2) or Tier 2 – Low risk outdoor cultivations (disturbed area greater than 43,560 ft^2; ~4,046 m^2), as defined by the Cannabis Policy (State Water Resources Control Board 2019a).

The fate of an indoor cannabis cultivation's industrial wastewater, including irrigation tailwater and brines resulting from water treatment or recycling, and the mechanism of that discharge, are important drivers for regulation by the SWRCB. Ultimately, all industrial wastewater from indoor cannabis cultivation activities is intended to end up in a permitted wastewater treatment facility, either by direct discharge to a publicly owned treatment works (POTW) via sewer or by hauling the wastewater off site. Prior to discharging or hauling industrial wastewater, indoor cultivation facilities may also use onsite water treatment systems. However, onsite treatment requires separate regulatory approval and may trigger additional monitoring and reporting requirements. Of active enrollees in the South Coast region with indoor cultivation, 489 (94%) discharge directly to a POTW, 23 (4%) haul wastewater offsite, and 4 (<1%) use some form of onsite treatment and/or water recycling to eliminate or minimize industrial wastewater. Discharge of industrial wastewater, including cannabis wastewater streams from indoor cultivation activities, to a septic system is considered a risk to groundwater quality and is prohibited without separate regulatory approval. Indoor facilities without a sewer connection have industrial wastewater and nonliquid cultivation related waste hauled off site to permitted waste facilities. The distribution of cultivators as a function of their mechanisms of wastewater discharge is shown on Figure 1.

In 2019, the SCRCU inspected 24 enrolled, actively cultivating indoor facilities, where SCRCU staff observed a wide range of cultivation practices and irrigation methods. Inspected indoor facilities ranged in size from a few hundred square feet (tens of square meters) with less than 100 plants to converted warehouses greater than 35,000 ft^2 (~3,251 m^2) with more than 6,500 plants. Twenty-one of the inspected facilities were using some type of hydroponic growing system. The hydroponic growing mediums observed were rockwool, coco fiber/chips, and perlite. All the facilities inspected used a municipal water source for their irrigation water. Observed irrigation methods included drip irrigation system, ebb-flow systems, flood tables, and hand watering. Water use was often not logged by cultivators and is highly variable dependent on the facility size and irrigation method. High efficiency irrigation methods such as drip irrigation systems, flood tables, and ebb-flow systems (Denver Public Health and Environment 2018) were observed at 14 of the inspected facilities. One of the larger facilities inspected, cultivating approximately 6,500 plants, provided the SCRCU with a water bill showing 372 hundred cubic feet or 278,256 gallons (~1 x 10^6 L) of water usage for one month of operation, though it should be noted that the water bill includes non-cultivation related water use and the cultivator's estimated water usage for cultivation was approximately 2,000 gallons (~7,570 L) per day. The source of the discrepancy between these two numbers is unknown. Of the inspected facilities, cultivators using ebb-flow systems or flood tables recycled their irrigation water. Facilities using automated drip-irrigation systems have little to no irrigation run-off, limiting their industrial wastewater stream to reverse osmosis brine and HVAC condensate. Indoor cultivation facilities often pretreat municipal source water before augmenting the treated water with nutrients for plant irrigation. Twenty of the 24 facilities treated the municipal source irrigation water prior to nutrification. Irrigation water pretreatment methods included reverse osmosis systems, ultraviolet light treatment, sediment filtration, and carbon filtration.

All inspected facilities were using carbon dioxide augmentation in the grow areas for various reasons including increased growth rates and biomass production (Poudel and Dunn 2017). Carbon dioxide concentrations of the air in the grow areas were observed between 200 and 1,900 parts per million depending on the growing stage, which is near the range typically encountered indoors (350-2,500 parts per million; Seppänen et al. 1999) and well below the Occupational Safety and Health Administration Permissible Exposure Limits of 5,000 parts per million.

Illicit Sites in the South Coast Region

Cultivation of cannabis beyond six plants and without applicable local and state permits or license is illegal in California (HSC §11358) and subject to additional violations, including violations of the California Food and Agriculture Code and California Water Code. The distribution of illicit cannabis cultivation sites inspected by the SCRCU as of 1 January 2020 is shown on Figure 2. The inspected illicit sites ranged in size from small “backyard” style grows to professional, industrial-scale operations. Some smaller sites were observed cultivating fewer than 100 plants in less than 185 m^2 mixed sunlight and artificial light greenhouses or in full sun outdoor areas. The largest site had greater than 31,000 plants, greater than 200 greenhouses, and disturbed an area of nearly 32,500 m^2.

Table 1 presents the site characteristics, water source, cultivation style, and irrigation method of 143 illicit cannabis cultivation sites where SCRCU staff performed inspections.

During site inspections the SCRCU documented observations of environmental impacts and potential environmental impacts, including the presence of onsite waterways as indicated in the National Hydrography Dataset (United States Geological Survey 2019), the presence of chemicals, site grading for cannabis cultivation, and substantial litter, and the threat of sediment mobilization. Table 2 presents the frequency of select observed and potential environmental impacts relevant to the 12 treatment and control categories outlined in the Cannabis Policy (State Water Resources Control Board 2019a).

The fate of irrigation tailwater and other cultivation related wastewaters was documented at each of the 143 sites inspected. Wastewater was observed discharged to the ground at 124 sites, to a waterway at 26 sites, to a lake or pond at three sites, and to a sewer at 10 sites. Some sites had multiple points of discharge.

DISCUSSION

Enrolled Cultivation Facilities in the South Coast Region

Distribution and style.—The distribution and style of cultivation, whether indoor, full sun outdoor, or mixed sunlight and artificial light greenhouse, in the South Coast region reflects local government restrictions to cultivation rather than logistic or geographic

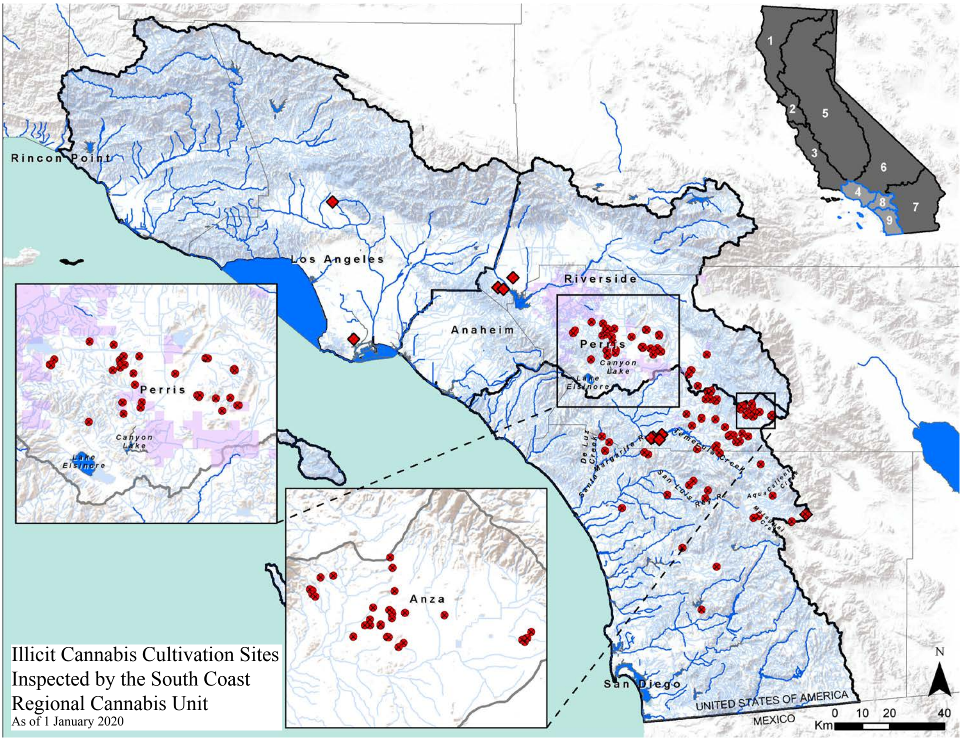


Figure 2. The SCRCU documented environmental impacts at 143 illicit cannabis sites inspected between 29 March 2018 and 1 January 2020. Sites that had outdoor cultivations, either full sun or mixed sunlight and artificial light greenhouses, are shown as red circles. Sites that were exclusively indoor cultivations are shown as red diamonds. Clean Water Act 303(d) (State Water Resources Control Board 2010) impaired waterbodies are indicated as bright blues on the map. National Hydrography Dataset (United States Geological Survey 2019) waterways are shown as light blues. The SCRCU identified 12 Clean Water Act 303(d) waterbodies, listed in Table 3, as having the potential to be threaten by illicit cannabis cultivation activities. In addition, the SCRCU identified 11 sites within Ground Water Protection Areas, shown in pink, as defined by the California Department of Pesticide Regulation (2020). These areas are vulnerable to leaching and runoff pesticide contamination from agricultural use. The SWRCB's nine regional waterboards are shown in the inset as defined in Figure 1.

constraints. Under MAUCRSA, cities and counties (for unincorporated areas) maintain the authority to establish ordinances that allow or prohibit commercial cannabis businesses within their jurisdictions. As of 1 January 2020, legal[2] commercial cannabis cultivation is limited to the following areas in the South Coast region:

- Santa Barbara County: Not applicable within South Coast region limits
- Ventura County: Port Hueneme
- Los Angeles County: Baldwin Park, Bell, Bellflower, Canoga Park, Carson, Commerce,

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[2] The list of counties and municipalities with ordinances allowing for commercial cultivation and the accompanying map (Figure 1) show the distribution of permitted cannabis cultivation along the southern coast of California as of 1 January 2020. This information is for informational purposes only. The authors make no warranty as to the accuracy of cannabis cultivation legality in the communities listed and shown in Figure 1. Readers are encouraged to contact their local or county government to determine the legality of commercial cannabis cultivation in their community.

Table 1. Characteristics of illicit cannabis cultivation sites inspected by the SCRCU in southern coastal California as of 1 January 2020.

Inspected Site Characteristics					
Water Source	Groundwater Well	Municipal	Diversion	Unknown	Not Applicable
	84	48	1	11	1[1]
Cultivation Style	Full Sun	Mixed Light[2]	Indoor		Not Applicable
	55	105	14		1[1]
Irrigation Method	Drip	Hand			Not Applicable
	77	90			1[1]

[1] No evidence of past or present cannabis cultivation was observed on the site.

[2] Mixed sunlight and artificial light greenhouses

Table 2. Select observed and potential environmental impacts relevant to the 12 treatment and control categories outlined in the Cannabis Policy (State Water Resources Control Board 2019a) observed at illicit cannabis cultivation sites inspected by the SCRCU in southern coastal California as of 1 January 2020.

Select Environmental Impacts	Present	Not Present	Not Applicable
Onsite Waterway	42	101	0
Non-EPA Registered Chemical	20	121	2[1]
Non-Approved Chemical for Cannabis	74	67	2[1]
Sediment Discharge Risk	133	10	0
Cannabis Cultivation Related Site Grading	71	72	0
Green Waste Litter	83	60	0
Domestic Waste Litter	97	46	0

[1] Either unknown due to a limited inspection of the site or no evidence of past or present cannabis cultivation was observed on the site.

Cudahy, Culver City, El Monte, Huntington Park, Long Beach, Los Angeles, Lynwood, Maywood, Montebello, Pasadena, Santa Monica, West Covina

- Orange County: Santa Ana
- San Bernardino County: Colton, San Bernardino
- Riverside County: Jurupa Valley, Lake Elsinore, Moreno Valley, Perris, San Jacinto, Unincorporated County[3]
- San Diego County: Chula Vista, La Mesa, Lemon Grove, Oceanside, San Diego

Currently, most cities in the South Coast region have ordinances banning commercial cannabis cultivation, but many cities are in various stages of drafting city ordinances that will allow commercial cannabis cultivation or are awaiting outcomes of November 2020

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[3] Commercial cannabis cultivation in unincorporated Riverside County is limited to specific zoned areas (County of Riverside 2018).

ballot measures to proceed with ordinance drafting. Other cities, such as Jurupa Valley and the City of Commerce have passed ordinances allowing commercial cannabis cultivation and have accepted commercial cannabis cultivation business applications but are in the process of reviewing and approving the businesses.

Of the counties within the South Coast region, only Riverside County has passed an ordinance permitting commercial cannabis cultivation. Riverside County Ordinance No. 348, effective 23 December 2018, allows for commercial cannabis cultivation within specified zones in unincorporated areas of the county (County of Riverside 2018). During the first year of implementation, up to 50 commercial cannabis cultivation businesses will be permitted for indoor artificial light or greenhouse mixed sunlight and artificial light cultivation (County of Riverside 2018). As of 1 January 2020, the county has approved 30 commercial cannabis cultivation facilities to proceed with the licensing process and permits are pending.

Enrollments in the South Coast region are concentrated in the city of Los Angeles, due in part to the preexisting medical cannabis industry permitted in the city since the passage of the Compassionate Use Act in 1996 (Proposition 215). With the passage of Proposition 64 and the adoption of MAUCRSA, the city established a three phase roll out to license commercial cannabis businesses. As of 1 January 2020, the city has rolled out phases 1 and 2, issuing licenses to existing (pre-MAUCRSA) cannabis businesses. Phase 3 of the city's licensing program will open the licensing window for new commercial cannabis businesses, and it is anticipated that the number of enrolled commercial cannabis cultivation facilities in the city will significantly increase after the closing of Phase 3.

The cultivation style of enrollees in the South Coast region reflects the urban nature of the areas where cultivation is permitted in southern California, with 99% of enrollees issued an indoor waiver of WDRs. This is also true of other predominately urban SWRCB regions. Within the San Francisco Bay RWQCB, 151 out of 177 (85%) enrollees were issued indoor waivers as of 1 January 2020. For comparison, statewide, 1,260 out of 4,391 (29%) active enrollments into the SWRCB's Cannabis General Order as of 1 January 2020 are indoor cultivations. Removing the SCRCU, the percentage of indoor cultivations drops to 745 out of 3,873 (19%) statewide. For the purpose of this comparison, we define northern California as the North Coast RWQCB, the San Francisco Bay RWQCB, and the northern geographic regions covered by the Central Valley RWQCB and the Lahontan RWQCB (enrollees with the Central Valley RWQCB's Redding and Rancho Cordova offices and Lahontan RWQCB's South Lake Tahoe office). In northern California, 396 out of 3,251 (12%) of active enrollments are indoor cultivations, and within the Emerald Triangle, less than 3% of nearly 2,500 enrolled cultivations with the North Coast RWQCB are indoor cultivations.

Water use.—Most of the South Coast region receives minimal rainfall and Los Angeles in particular imports approximately 86% of their water from either the Eastern Sierra Nevada via the Los Angeles Aqueduct, the Sacramento Bay Delta via the California Aqueduct, or the Colorado River via the Colorado River Aqueduct (Los Angeles Department of Water and Power 2019). Recent multi-year droughts have led water providers to enact mandatory conservation measures. Water consumption concerns are not unique to the cannabis industry and industry specific water use restrictions are not established (Denver Public Health and Environment 2018). However, efforts to reduce water consumption were observed in many of the inspected permitted facilities. This proactive approach to efficient water use is mutually beneficial to the cultivator, in terms of lower operational costs, and to the environment.

Discharge.—Indoor cannabis cultivations that discharge wastewater to a community sewer system pose a low threat to water quality, provided that the discharges are compliant with the sewer agency's requirements. Although cannabis cultivation effluent may contain nutrients such as phosphate or nitrate, salinity constituents such as sodium, chloride, potassium, calcium, sulfate, and magnesium, or other constituents such as iron, manganese, molybdenum, boron, and silver (State Water Resources Control Board 2019a), as of 1 January 2020, local POTWs in the South Coast region do not have pollutant limits, pre-treatment requirements, or discharge permits specifically for commercial cannabis operations. In November of 2019, the SCRCU began a cultivation effluent sampling project to characterize wastewater concentrations originating from indoor cannabis cultivations. Preliminary analytical results from two facilities in the city of Los Angeles show pollutant concentrations well below established local limits for industrial wastewater. Full results of the effluent sampling project will be reported at a future date.

Illicit Sites in the South Coast Region

Legalization does not eliminate the need for enforcement of illicit activities, particularly illicit outdoor cannabis cultivations which pose the highest risk to the environment and water quality. Polson and Petersen-Rockney (2019) state that "California's cannabis legalization has enabled a kind of multi-agency neoprohibitionism at the county level, one that reinforces older criminal responses with new civil-administrative strategies and authorities". While this statement accurately identifies the control that local jurisdictions have on the emerging cannabis market, which is particularly limiting for cultivation, it does not consider the economic complexities of a regulated market structure and the pervasive implications of illicit activities, including their environmental impact. An important component to ensure the success of the regulated market structure is prevention of black-market undercutting, not unlike other regulated industries, and the SCRCU has been encouraged to pursue enforcement action against illicit cultivators by licensed cultivators operating within the legal market. However, among the multiple agencies that regulate and/or take part in cannabis related enforcement activities, the primary mission is specifically non-economics based, but rather is meant to directly address environmental impacts. Whereas enrolled/permitted cultivation sites must adhere to requirements that protect water quality to maintain their permitted status, unregulated/unpermitted cannabis cultivation sites are not properly engineered and rarely implement best management practices that would prevent environmental degradation and minimize potential water quality impacts. The SWRCB's strategy for enforcement is to, "ensure that the diversion of water and discharge of waste associated with cannabis cultivation does not have a negative impact on water quality, aquatic habitat, riparian habitat, wetlands, or springs", which includes action for noncompliance with the Cannabis General Order at enrolled sites and enforcement of the California Water Code on illicit cultivation activities (State Water Resources Control Board 2019a). In addition, enforcement action is given to Priority Watersheds which are designated for their "high value water quality, water supply, and habitat factors or current impairments to these factors" (State Water Resources Control Board 2019c). "Priority Watershed are at increased risk of environmental impacts due to the potential of cannabis cultivation activities to severely harm important watershed features and further aggravate existing impairments" and include Clean Water Act 303(d) listed waterbodies (State Water Resources Control Board 2010) and RWQCB priority designations (State Water Resources Control Board 2019c).

Clean Water Act impaired waterbodies.—During the 143 inspections conducted by the SCRCU within the South Coast region, 12 Clean Water Act 303(d) impaired waterbodies were identified as potentially threatened by illicit cannabis cultivation activities. Table 3 identifies these waterbodies and their impairments. Discharges from cannabis cultivation activities have the potential to aggravate existing impairments, particularly impairments from nutrients, nitrogen, phosphorus, and sulfates, total dissolved solids, pH, and toxicity based impairments resulting from harmful algae blooms (State Water Resources Control Board 2019a) which imply a nutrient imbalance.

San Diego RWQCB priority designations.—The semiautonomous RWQCBs designate priority water systems based on the local conditions within their regions. The San Diego RWQCB identified key areas for priority beneficial uses within their Practical Vision strategic plan (San Diego Regional Water Quality Control Board 2013), as well as identified high quality waters through the Clean Water Act 305(b) Report (San Diego Regional Water Quality Control Board 2016). High Quality and High Priority (HQHP) stream systems were identified as areas of special importance for the Habitat and Ecosystem Key Beneficial Uses, most critical to protecting human and environmental health (San Diego Regional Water Quality Control Board 2016, 2017). These HQHP stream systems also include refer-

Table 3. Clean Water Act 303(d) waterbodies identified as having the potential for impact by illicit cannabis cultivation activities at sites inspected by the SCRCU as of 1 January 2020.

Clean Water Act 303(d) Waterbodies and Impairments							
Impaired Waterbody	**Nutrients**	**Total Nitrogen as N**	**Nitrogen**	**Phosphorus**	**Sulfates**	**TDS**[1]	**Other(s)**
Canyon Lake	X						Pathogens
Cloverdale Creek				X		X	
De Luz Creek			X		X		Iron, Manganese
Gopher Creek							Benthic Community Effects
Keys Creek							Selenium
Rainbow Creek			X	X	X	X	Iron
San Dieguito			X	X		X	Enterococcus, Fecal Coliform
Santa Gertudis Creek				X			Chlorpyrifos, Copper, E. Coli, Fecal Coliform, Iron, Manganese
San Luis Rey River, Upper		X					
Santa Margarita River, Upper				X			
San Vicente Reservoir		X			X		Chloride, Color, pH (High)
Temecula Creek				X		X	Chlorpyrifos, Copper

[1]Total Dissolved Solids (TDS)

ence stream segments identified as being minimally disturbed by human activity (Ode et al. 2016). A total of 10 illicit cannabis cultivation sites were identified and inspected within the following HQHP stream systems, four sites within the De Luz Creek system, one site in Arroyo Seco Creek system, two sites within the Pauma Creek-San Luis Rey River system, one within the Agua Caliente Creek system, and two within the Matagual Creek-San Luis Rey River system.

Santa Ana RWQCB priority designations.—The Santa Ana RWQCB Basin Plan (Santa Ana Regional Water Quality Control Board 2019) incorporates a salt and nutrient management plan that identifies nitrate and salinity concentrations in groundwater and surface water as key metrics for compliance with the state's Antidegradation Policy (Mulligan 1968) in the Santa Ana River and San Jacinto River watersheds. The Basin Plan reports that the most serious threat to water quality in both the Santa Ana River and San Jacinto River watershed basins is the build-up of nutrients and dissolved minerals, or salts, in the groundwater and surface waters (Santa Ana Regional Water Quality Control Board 2019). Likewise, the Basin Plan calls for controls on salt loadings from all water uses including residential, commercial, industrial, and agricultural (Santa Ana Regional Water Quality Control Board 2019). The application of fertilizers and the irrigation of agricultural crops is identified as one of the principal causes of the salt problem in the Santa Ana Region. Implementation of a salt and nutrient management plan requires that discharges of waste be regulated to meet objectives for the concentrations of salinity, expressed as total dissolved solids, and nitrate, expressed as nitrogen, in groundwater Management Zones of the Santa Ana Region shown in Figure 3 (Santa Ana Regional Water Quality Control Board 2019). The Cannabis General Order (State Water Resources Control Board 2019b) requires Tier 2 cultivations with aggregate cultivation areas greater than one acre (~4,046 m^2) to provide a Nitrogen Management Plan to minimize discharge and protect water quality. Six illicit cannabis cultivation sites, discharging their irrigation tailwater to the ground, were identified and inspected within a groundwater Management Zone in the San Jacinto River watershed. Without a Nitrogen Management Plan to manage and minimize discharges of total dissolved solids and nitrates, these illicit cannabis sites are likely to contribute to the degradation of the San Jacinto River watershed.

Pesticide use.—Pesticide use on permitted legal cannabis cultivation sites is regulated by the California Department of Pesticide Regulation (DPR 2017), and the Bureau of Cannabis Control established action levels for residual pesticides on cannabis products heading to the market (16 California Code of Regulations (CCR) §5719). Greater than 50% of the inspected illicit cultivation sites had pesticides not approved for use on cannabis. The observed pesticides included Federal Insecticide, Fungicide, and Rodenticide Act Restricted Use Pesticides (7 USC §136 et seq.), California Restricted Materials (3 CCR §6400), pesticides on the California Groundwater Protection List (3 CCR §6800), and pesticides not registered for food use. Pesticides not registered for use by the United States Environmental Protection Agency (US EPA) were observed on 20 (14%) inspected sites. These are pesticides that are not legally allowed for distribution and use in the United States. Active ingredients in those pesticides included carbofuran, methamidophos, zinc phosphide, chlorothalonil, metalaxyl-m, and chlorantraniliprole. On sites with carbofuran, dead wildlife was sometimes observed, typically dead rodents or birds with dead insects nearby that had been exposed to the pesticide. Due to a lack of resources to remove these hazardous materials many of the pesticides are left on site and remain a potential risk to human health and the environment while civil and/or criminal enforcement is pending.

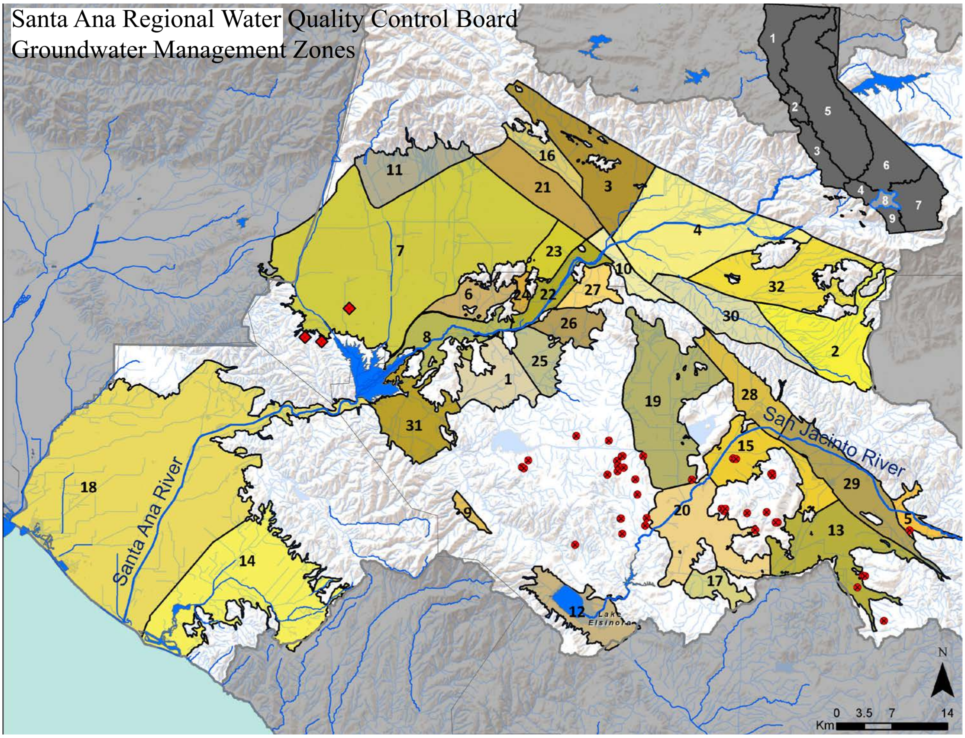

Figure 3. The Santa Ana RWQCB Basin Plan (Santa Ana Regional Water Quality Control Board 2019) identifies 32 groundwater Management Zones in the Santa Ana River and San Jacinto River watersheds where salt and nutrient water quality objectives protect beneficial uses. The SCRCU documented environmental impacts at 47 illicit outdoor cannabis cultivation sites within the Santa Ana Region between 29 March 2018 and 1 January 2020. Sites that had outdoor cultivations, either full sun or mixed sunlight and artificial light greenhouses, are shown as red circles. Sites that were exclusively indoor cultivations are shown as red diamonds. Of those 47 sites, six illicit cannabis cultivation sites were overlying groundwater Management Zones in the San Jacinto River watershed and had discharged irrigation wastewater to the ground. The SWRCB's nine regional waterboards are shown in the inset as defined in Figure 1. The 32 groundwater Management Zones are defined as follows: Arlington – 1, Beaumont – 2, Bunker Hill-A – 3, Bunker Hill-B – 4, Canyon – 5, Chino-East - 6, Chino-North – 7, Chino-South – 8, Coldwater – 9, Colton – 10, Cucamonga – 11, Elsinore – 12, Hemet-South – 13, Irvine – 14, Lakeview/Hemet-North – 15, Lytle – 16, Menifee – 17, Orange County – 18, Perris-North – 19, Perris-South – 20, Rialto – 21, Riverside-A – 22, Riverside-B – 23, Riverside-C – 24, Riverside-D – 25, Riverside-E – 26, Riverside-F – 27, San Jacinto-Lower Pressure – 28, San Jacinto-Upper Pressure – 29, San Timoteo - 30, Temescal - 31, Yucaipa – 32.

The DPR began addressing pesticide contamination in groundwater and drinking water supplies from agricultural use with the passage of the Pesticide Contamination Prevention Act. The DPR established a Groundwater Protection Program (2020) that evaluates use restrictions on pesticides that have been detected in groundwater and have the potential to contaminate groundwater, such as pesticides on the Groundwater Protection List (3 CCR §6800). The DPR also defined Ground Water Protection Areas (GWPAs) which are areas vulnerable to pesticide contamination from agricultural use. The DPR identified leaching and runoff as pathways or mechanisms for pesticides to enter the groundwater system (2020). Illicit cultivation sites without engineering controls and best management practices to prevent discharge to the ground and runoff of pesticides and wastewater containing pesticides, pose a risk to GWPAs. Of the 74 illicit sites identified as having non-approved chemicals for cannabis cultivation on site, 11 sites were within GWPAs and 44 sites were within 5 kms of a GWPA (Figure 2).

Sediment mobilization.—Sediment, both clean native soils and impacted soils, can be a significant source of water pollution and degradation of water quality (Santa Ana Regional Water Quality Control Board 2018). The Cannabis General Order (State Water Resources Control Board 2019b) requires Tier 1 and Tier 2 Moderate risk cultivators to provide a Site Erosion and Sediment Control Plan to minimize the discharge of sediment; higher risk sites are also required to provide a Disturbed Area Stabilization Plan prior to approval to begin cultivation. Uncontrolled mobilization of sediment has the potential to alter the landscape, change water flow patterns, and lead to habitat loss and/or fragmentation. Unpermitted site development and grading may not consider or account for the cumulative effects of sediment mobilization. Site grading, for the purpose of illicit cannabis cultivation, was observed at nearly half of the sites inspected. In multiple cases, grading had occurred within waterways and riparian environments. Evidence of sediment mobilization or the potential for sediment mobilization was observed at all mixed sunlight and artificial light greenhouse and full sun outdoor sites inspected, 133 out of 143 sites. Erosion channels and rills and fluid mobilized growing medium and potting soil was ubiquitous. Stockpiles of soil were typically stored on the bare ground without engineering controls and containment to prevent mobilization. Without engineering controls and the application of best management practices, full sun outdoor and mixed sunlight and artificial light greenhouse cultivation sites are potentially significant sources of sediment and threats to water quality. One of the larger and more environmentally egregious inspected sites was located in a HQHP stream system in San Diego County. The site was set on slopes greater than 40 degrees that had been cut and graded for greenhouses, vegetation from the slope side was removed, and no engineering controls or best management practices were present to prevent sediment mobilization or cultivation runoff. The disturbed area was estimated to be approximately 32,500 m^2. An intermittent stream, tributary to an impaired waterway for nitrogen (State Water Resources Control Board 2010), was located approximately 60 m downslope. Rills and erosion channels were observed throughout the site and discharge of sediment and cultivation runoff was observed downslope toward the stream.

Litter.—Perhaps the most visible impact to the environment from illicit sites is litter. Extensive litter was observed on illicit cultivation sites and included cultivation related litter such as plastic irrigation lines, chemical containers with residue, and plastic sheeting or mesh, and domestic litter such as food containers, fuel and petroleum products, paper, cans and bottles, and miscellaneous household products. Litter was observed intentionally disposed of in ditches, waterways, and excavated pits. Living conditions often resulted in discharges of raw untreated sewage onto the ground, into pits, or directed intentionally into waterways. Pit toilets and intentional discharge of sewage from recreation vehicles via pipes or hoses were also observed. The deleterious effects of litter in waterways impacts beneficial uses and poses a threat to aquatic life, wildlife, and public health (State Water Resources Control Board 2015). The impacts of litter on the marine and terrestrial environment are well documented, and beyond the deterioration of aesthetics, the extensive litter observed threatens ingestion and entanglement by fish and wildlife, habitat alteration and degradation that affects migration, spawning, reproduction, and development of organisms, and poses a threat to human health (State Water Resources Control Board 2015).

Surface water diversion.—Lack of precipitation and reliable surface water make diversion of water rare in the South Coast region. Most of the water ways in the South Coast region are intermittent or ephemeral and dry most of the year. As a result, water diversion was observed on only one site, located on the southern flank of Palomar mountain at an elevation of approximately 1,050 m above sea level.

Conclusions

During the first two years of legal recreational cannabis cultivation in California, the SWRCB's Cannabis Cultivation Program enrolled 4,391 cultivators into the Cannabis General Order. The SCRCU, covering approximately 28,500 km^2 in southern coastal California, account for 519 of those cultivators, 516 of which are indoor cultivations. The risk to water quality associated with discharges of waste from indoor cultivations is minimal within the guidelines of the Cannabis Policy (State Water Resources Control Board 2019a), especially when considering that those cultivators are discharging their industrial process wastewater to a permitted, and further regulated, POTW or wastewater treatment facility. As a result, all but three outdoor cultivators enrolled with the SWRCB in the SCRCU were issued waivers of WDRs as of 1 January 2020.

Enforcement of illicit cultivation for the purpose of the protection of water quality and habitat also plays an important role in the SCRCU's Cannabis Cultivation Program. The SCRCU inspected 143 illicit cultivation sites as of 1 January 2020. Illicit cultivation sites have the potential to pose a significant risk to the environment. Observations from inspections of illicit sites identified threats to Clean Water Act 303(d) waterbodies and RWQCB priority water systems, and observed illegal and/or unauthorized pesticide use, evidence of and potential for sediment mobilization, and ubiquitous and substantial litter. These illicit cultivation sites pose a threat to water quality, aquatic and riparian habitats, and fish and wildlife, though the effects have not yet been quantified.

Limitations

The authors make no warranty as to the accuracy of cannabis cultivation legality in the communities discussed. Readers are encouraged to contact their local or county government to determine the legality of commercial cannabis cultivation in their community.

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AUTHOR CONTRIBUTIONS

Conceived and designed the study: BMC, CSP, ETL, PY, MAZ
Collected the data: BMC, CSP, ETL, PY, MAZ
Performed the analysis of the data: BMC, CSP, ETL, PY, MAZ
Authored the manuscript: BMC, CSP, ETL
Provided critical revision of the manuscript: BMC, CSP, ETL, PY, MAZ

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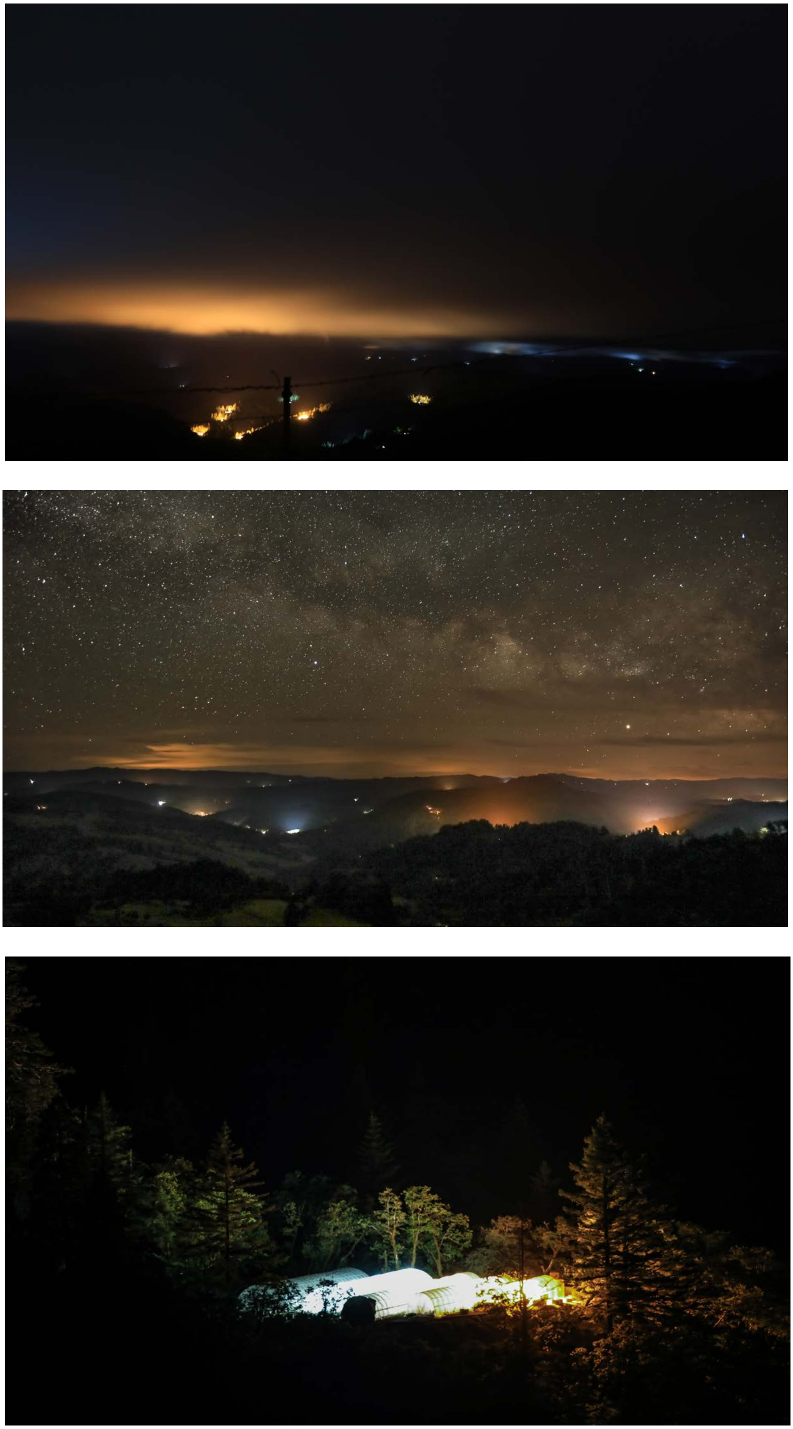

Examples of light pollution from cannabis cultivation operations in southern Humboldt County, 2018–2020. Photo credit: LoMaX

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A review of the potential impacts of artificial lights on fish and wildlife and how this may apply to cannabis cultivation

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Artificial lighting is used at cannabis cultivation sites in California to promote yield, for task lighting, and to provide security. While our understanding of how fish and wildlife respond to the artificial lights associated with cannabis cultivation specifically is in its infancy, studies assessing species' responses to other forms of artificial lighting at night have been ongoing for decades. We provide a review of these studies, with the goal of illuminating how artificial lights may influence the activity, movement, navigation, migration, phenology, and physiology of fish and wildlife populations.

Key words: activity patterns, artificial light, cannabis, fish, migration, movement, phenology, physiology, wildlife, photobiology

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Light plays a vital role in ecosystems by functioning as both an energy and an information source (Gaston et al. 2012, 2013). The addition of artificial light (i.e., light produced by humans) into a landscape can disrupt this role, altering the natural diel, lunar, and seasonal cycles under which species have evolved. This can influence a broad range of system processes including primary productivity in plants, wildlife activity patterns, species interactions, availability and detectability of food resources, movement and migration, timing of phenological events, and physiological functions (Longcore and Rich 2004, Da Silva et al. 2015, Bliss-Ketchum et al. 2016, Spoelstra et al. 2017). Further, because of sky glow (i.e., scattered light in the atmosphere), the reach of artificial light can extend far beyond the area that is directly illuminated (Longcore and Rich 2004). On cloudy nights in urban and suburban areas, for example, the sky glow effect can be of an equivalent or greater magnitude than high-elevation summer moonlight (Kyba and Hölker 2013).

Artificial lighting is increasingly being used in indoor and mixed-light (i.e., greenhouse) cannabis cultivation to promote yield, and for security around the perimeter of cannabis cultivation sites (CDFA 2017). While understanding how fish and wildlife respond to the artificial lights associated with cannabis cultivation, specifically, is an emerging question, studies aimed at assessing species' responses to other forms of artificial light have been ongoing for decades (Rowan 1929, Lashbrook and Livezey 1970, Pedersen and Larsen 1982, Thorpe 1987). Prior knowledge of how artificial light influences fish and wildlife species led the California Department of Food and Agriculture (CDFA), the primary commercial cannabis licensing authority, to require cannabis cultivation licensees to comply with several environmental protection measures pertaining to artificial light (CDFA 2017). These include ensuring that all outdoor lighting used for security purposes is shielded and downward facing, and that all lights used for cultivation are shielded from sunset to sunrise to avoid nighttime glare (CDFA 2017). To elucidate why these protective measures were put into place, and to predict how artificial lights associated with cannabis cultivation may influence fish and wildlife species across California, we review prior studies that assessed the influence of artificial light on species' 1) activity patterns, 2) movement, navigation, and migration, and 3) phenology and physiology. In this paper, we review these potential impacts to fish and wildlife resources, as well as potential approaches for mitigating the impacts of artificial lights.

ACTIVITY PATTERNS

Artificial light improves diurnal and crepuscular species' ability to see at night, allowing them to extend their period of activity into hours of natural darkness (Boujard and Leatherland 1992; Longcore and Rich 2004; Gaston et al. 2013). This activity pattern shift has been predominantly documented in birds, with bird species like the American Robin (*Turdus migratoriusi*) and Northern Mockingbird (*Mimus polyglottos*) beginning morning choruses earlier in the dawn and earlier in the year in areas with artificial lights (Table 1; Derrickson 1988; Miller 2006). For some species, this effect was found to be strongest at higher light intensities (Da Dilva et al. 2014, 2015). Diurnal bird species and salmonid fishes such as Lesser Kestrels (*Falco naumanni*), plover species, European Robins (*Erithacus rubecula*), Blue Tits (*Cyanistes caeruleus*), Rainbow Trout (*Oncorhynchus mykiss*), and Atlantic Salmon (*Salmo salar*) (Boujard and Leatherland 1992; Boeuf and Le Bail 1999; Negro et al. 2000; Santos et al. 2010; Byrkjedal et al. 2012), have also been shown to extend their foraging activities into the night in artificially illuminated areas. In the salmonid species, this extended feeding period led to increased growth rates and muscle mass (Boujard and Leatherland 1992; Boeuf and Le Bail 1999).

Conversely, other species may have reduced foraging success or reduced nighttime activity in artificially illuminated environments (Vogel and Beauchamp 1999; Gaston et al. 2013). Prey detection in some drift feeding and piscivorous fish species, for example, is dependent on the contrast between prey and background lighting. Consequently, these species exhibit greater visual sensitivity under low light conditions, and their ability to detect prey may be reduced in artificially lit environments (Tanaka 1970; Blaxter 1975). Artificial night lighting has also been found to impact juvenile salmonid overwintering success by delaying the emergence of salmonids from benthic refugia and reducing their ability to feed during the winter (Contor and Griffith 1995; Bradford and Higgins 2001).

A species may reduce their nighttime activity, alternatively, if their vulnerability to predation increases in brighter conditions (Navara and Nelson 2007; Gaston et al. 2013).

Table 1. Examples of changes in activity patterns resulting from artificial nighttime lighting.

Taxa	Species	Response	Source
Birds	Common Blackbird (*Turdus merula*)	Birds residing in the city started their activity before civil twilight, whereas birds residing in a rural forest synchronized their activity to the onset of twilight.	Dominoni et al. 2013
	American Robin (*Turdus migratoriusi*)	Chorus initiation time, relative to twilight, was positively correlated with the amount of artificial light. Mean high, intermediate, and low light levels were 3.91, 1.26, and 0.00 lx, respectively.	Miller 2006
	Northern Mockingbirds (*Mimus polyglottos*)	In an arboretum, mated males rarely sang at night under natural lighting conditions. This pattern was disrupted in more developed areas with artificial lighting.	Derrickson 1988
	Barred Owl (*Strix varia*), Long-Eared Owl (*Asio otus*)	In a laboratory setting, owls detected and attacked mice directly at light intensities comparable to weak daylight and the light given by a full moon. At dim light intensities, conversely, owls were more likely to have to search on foot.	Dice 1945
Mammals	Deer mice (*Peromyscus maniculatus*)	In an indoor habitat chamber, mice decreased their total activity as moonlight increased and were more active in areas with cover. Light bulbs were used to simulate moonlight, with 0.05, 0.17, 0.35, 0.93, and 2.20 lx representing new, quarter, half, three-quarter, and full moons, respectively.	Falkenberg and Clarke 1998
	Merriam's kangaroo rat (*Dipodomys merriami*)	During full moons, rats were more likely to be found in their day burrows and when they emerged, they stayed closer to home. This suppression of nocturnal activity led to an increase in crepuscular activity.	Daly et al. 1992
	Merriam's kangaroo rat	In large outdoor enclosures, rats reduced their foraging in the presence of artificial lighting (i.e., six floodlights set to approximate the light intensity of a full moon).	Brown et al. 1988
	Snowshoe hare (*Lepus americanus*)	Winter track transects showed that hares were less active in open areas during moonlit nights when compared with dark nights.	Gilbert and Boutin 1991

Table 1 continued.

Taxa	Species	Response	Source
Herpetofauna	Prairie Rattlesnake (*Crotalus viridis*)	In a captive setting, light bulbs were used to simulate moonlight, with 0.06, 0.35, 1.00, and 2.10 lx representing new, half, three-quarter, and full moons, respectively. Nocturnal snake activity was greater in new moonlight when compared to three-quarter and full moonlight. Snakes also spent proportionally more time in open areas on darker nights.	Clarke et al. 1996
	Common Toad (*Bufo bufo*)	The toad uses visual cues for prey detection, even at very low light intensities. Light only becomes a limiting factor if it falls below 30.00 µlx (i.e., 30 millionths of a lx).	Pedersen and Larsen 1982
	Long-Nosed Snake (*Rhinocheilus lecontei*)	The snake's small mammal prey has decreased due to light pollution, which may be responsible for the decline of this nocturnal species.	Perry and Fisher 2006
Invertebrates	Aquatic invertebrate (*Daphnia retrocurva*)	In an enclosure experiment, movement was significantly greater in amplitude (2 m higher) and magnitude (10-20% more individuals) in black enclosures that blocked 96% of irradiance vs. open enclosures that had light pollution levels less than that of full moonlight.	Moore et al. 2000
	Drifting aquatic insects	Insects were 37% less abundant at artificially lit sites (mean = 1482.00 lx) than at ambient sites (< 2.00 lx). The effects of light were most notable in large streams, which had a 58% decrease in Simuliidae and 51% decrease in Baetidae (when compared to sites under ambient conditions).	Henn et al. 2014
Fish	Rainbow Trout (*Oncorhynchus mykiss*)	In a laboratory setting, fish exposed to a 24L:0D continuous light cycle had a significantly higher daily growth rate than other experimental photoperiod groups.	Türker and Yildirim 2011
	Rainbow Trout	Trout reaction distance to prey species increased as light levels exceeded 5.00 lx, reaching an observed distance threshold or saturation effect (SIT) around 20.00 lx.	Mazur and Beauchamp 2003
	Rainbow Trout	Prey detection capability decreased when light levels exceeded the SIT (17.80 lx) because visual prey detection is dependent on the contrast between their prey and the background lighting.	Vogel and Beauchamp 1999

This pattern appears to be widespread, having been documented in species ranging from small mammals to snakes to amphibians to invertebrates (Table 1). Insectivorous bat species in Europe, including the lesser horseshoe bat (*Rhinolophus hipposideros*), Geoffroy's bat (*Myotis emarginatus*), and lesser mouse-eared bat (*M. oxygnathus*), for example, showed significantly decreased activity and/or a delay in the start of commuting behavior when exposed to light, likely as a predator avoidance strategy (Stone et al. 1999; Boldogh et al. 2007; Spoelstra et al. 2017).

One species altering its activity patterns due to artificial light can have cascading impacts on numerous other animals, including the species' predators, competitors, and prey. If prey species reduce their nighttime activity in areas with artificial light, for example, it can make prey detection harder and increase the energy demands of the respective predator (Table 1; Buchanan 1993). Alternatively, if prey species are attracted to artificial light, it can make prey detection easier and may result in changes in the movement patterns or distributions of the species' predators (Longcore and Rich 2004; Becker et al. 2013; Gaston et al. 2013). Artificial light tends to attract insects from the orders Lepidoptera, Diptera, Trichoptera, Hemiptera, Coleoptera, and Hymenoptera, for example, which then attract insectivorous bird and bat predators (Table 1; Santos et al. 2010; Longcore et al. 2015; Minnaar et al. 2015; Spoelstra et al. 2017; Welbers et al. 2017). Lastly, artificial light may make prey detection easier for predators that rely on visual cues to locate prey, as has been found with certain species of owls, toads, and salmon (Table 1; Dice 1945; Pedersen and Larsen 1982; Mazur and Beauchamp 2003).

MOVEMENT, NAVIGATION, AND MIGRATION

Artificial light can function as a barrier to connectivity, which may contribute to isolated populations, reduced genetic diversity, increased species' susceptibility to disease, and limited access to resources (Table 2; Bliss-Ketchum et al. 2016). Some mammal species, for example, are less likely to use road under-crossings that are illuminated when compared to those that are dark (Bliss-Ketchum et al. 2016). *Plecotus* and *Myotis* bat species in the Netherlands, alternatively, avoided areas that were illuminated by white or green light, resulting in the loss of these areas as potential habitat (Spoelstra et al. 2017).

By masking the natural light signals (e.g., through sky glow) that guide species' movements, artificial light can also have major disruptive effects on navigation and migration patterns in a variety of species (Table 2; Rowan 1932; Lowe 1952; Gaston et al. 2013; Bennie et al. 2015). In Pacific salmon (*Oncorhynchus* spp.), for instance, adult migrations and the out-migration of juveniles can be slowed or halted by the presence of artificial lights (Tabor et al. 2004; Nightingale et al. 2006). Similarly, the orientation of nocturnally migrating birds, the homing behavior of Red-Spotted Newts (*Notophthlamus viridescens*), and the vertical migration of larval salamanders (*Ambystoma* spp.), have all been documented to be disrupted by artificial light (Anderson and Graham 1967; Phillips and Borland 1992, 1994; Poot et al. 2008).

PHENOLOGY AND PHYSIOLOGY

Light mediates species' input and interpretation of day length, which can affect the output of certain hormones that regulate physiological events like development, reproduction, hibernation, dormancy, smoltification, and migration (Table 3; Hoffnagle and Fiviz-

Table 2. Examples of altered animal movement or migration patterns associated with artificial nighttime lighting.

Taxa	Species	Response	Source
Mammals	Black-tailed deer (*Odocoileus hemionus columbianus*)	Deer showed sensitivity even to nearby lights, using under-road crossings less often when neighboring sections were lit (high = 172.00 lx; low = 54.00 lx) compared to when none were lit.	Bliss-Ketchum et al. 2016
	Deer mice (*Peromyscus maniculatus*), opossum (*Didelphis virginiana*)	Mice and opossum used under-road crossings significantly more often in ambient conditions than in lit (high = 172.00 lx; low = 54.00 lx).	Bliss-Ketchum et al. 2016
	California bat species	Artificial lights can disturb roosting bats and potentially lead to the abandonment of maternity roosts.	Johnston et al. 2004
Herpetofauna	Common Toad (*Bufo bufo*)	During mass emigration of newly metamorphosed toads away from their aquatic environment, more toads aggregated in areas illuminated by streetlamps than in unlit areas.	Baker 1990
	Larval salamanders (*Ambystoma* spp.)	Vertical migration, which is associated with feeding, was correlated with decreased light intensity.	Anderson and Graham 1967
Invertebrates	Monarch Butterfly (*Danaus plexippus*)	In a lab, Monarch's circadian clock was disrupted when exposed to constant light, likely because they rely on light cues to migrate. Further, they were unable to orient in the correct direction when exposed to advanced light (i.e., sun compass had been advanced by 6 hours).	Froy et al. 2003
Fish	Juvenile Pacific salmonids (*Oncorhynchus sp.*)	Salmon fry migrated downstream at a slower rate under higher light intensities (1.08 and 5.40 lx) than under complete darkness (0.00 lx).	Tabor et al. 2004
	Rainbow Trout (*Oncorhynchus mykiss*) and Atlantic Salmon (*Salmo salar*)	Locomotor activity of salmonids is strongly influenced by an endogenous circadian clock entrained to 12L:12D cycles.	Iigo and Tibata 1997, Richardson and McCleave 1974, Thorpe 1987
	General	Input of artificial light increases species abundance by attracting fish to light sources, potentially concentrating predator and prey fish species. This can cause unnatural top-down regulation of fish populations.	Nightingale et al. 2006, Becker et al. 2013

Table 3. Examples of phenological and physiological changes associated with artificial nighttime lighting.

Taxa	Species	Response	Source
Mammals	Social Voles (*Microtis socialis*)	In a lab setting, night-light pulses (i.e., three 15 min 450 lx light pulses) negatively affected winter acclimization of thermoregulatory mechanisms.	Zubidat et al. 2007
	Deer mice (*Peromyscus maniculatus*)	In a lab setting, mice maintained in long day lengths (16L:8D) were more likely to develop induced tumors when compared to animals maintained in short day lengths (8L:16D); exposure to constant dim light (0.21 lx) significantly increased the growth rate of induced tumors.	Navara and Nelson 2007
Birds	Juncos (*Junco* spp.)	In a lab setting, juncos exposed to artificial light came into reproductive condition out of season.	Rowan 1925
	Common Blackbird (*Turdus merula*)	Nocturnal illumination (0.30 lx) suppressed reproductive activity, which demonstrates that chronic low intensities of light at night can affect the reproductive system.	Dominoni et al. 2013
Herpetofauna	Northern Green Frog (*Rana clamitans melanota*)	Males produced fewer advertisement calls and moved more frequently when illuminated by a maglite flashlight (approximately 1 m surrounding the frog was illuminated), which has the potential to reduce recruitment rates.	Baker and Richardson 2006
	Northern Cricket Frogs (*Acris crepitans*), American Bullfrog (*Lithobates catesbeianus*), Rio Grande Leopard Frog (*L. berlandieri*)	Fewer frogs called when the site was illuminated by a hand-held high-intensity spotlight (~ 38.00 lx at 5 m and 9.00 lx at 10 m) than when kept dark. Frogs also called less intensely during lit conditions vs. unlit.	Longcore and Rich 2004, Hall 2016
	American Toad (*Bufo americanus*)	Toads use photoperiod cues to behaviorally thermoregulate.	Beiswenger 1977
	Western Fence Lizard (*Sceloporus occidentalis*)	In a lab setting, photoperiod influenced the lizard's critical maximum temperature and ability to maintain or depress subdermal temperatures.	Lashbrook and Livezey 1970

Table 3. continued.

Taxa	Species	Response	Source
	Tiger Salamander (*Ambystoma tigrinum*)	Under constant light, the salamander's production of melatonin was disrupted, which altered metabolic rates and required the animal to spend more time foraging.	Perry et al. 2008
Invertebrates	Beetles (family Lampyridae)	Artificial lights may be contributing to the decline of these beetles given they rely on bioluminescence for mate location.	Lloyd 2006
Fish	Chinook Salmon (*Oncorhynchus tshawytscha*)	In an experimental setting, exposure to unnatural photoperiods (24L:0D and 9L:15D) delayed smoltification and decreased body condition when compared to salmon raised under natural light conditions.	Hoffnagle and Fivizzani 1998
	Rainbow Trout (*Oncorhynchus mykiss*)	Fish exposed to artificial photoperiods (14 days at 10L:14D, 60 days at 24L:0D, 30 days at 10L:14D, and 30 days at 12L:12D) experienced immune suppression, and increased levels of cortisol that is indicative of stress.	Leonardi and Klempau 2003
	Rainbow Trout	Exposure to constant photoperiod (24L:0D), created by artificial lighting in a lab setting, advanced spawning timing by 6 weeks to 3 months, and induced a repeat 4-7 month spawning cycle in those fish.	Bromage et al. 1984, Elliot and Bromage 1984
	Rainbow Trout	In a lab setting, alteration of daylength by artificial light manipulation (24L:0D) may indirectly modify growth rate (up to 25%) by increasing food intake and modify muscle mass by increasing exercise.	Taylor et al. 2006

zani 1998; Bradshaw and Holzapfel 2010). Further, photoperiodic control allows species to synchronize reproductive activities and to coordinate key life cycle events with suitable weather conditions (Gaston et al. 2013). When natural photoperiods are disrupted by artificial lights, species may become asynchronous with climatic and environmental conditions (e.g., mismatched reproductive activity with new plant growth or the reproductive activity of prey), which in turn, may negatively impact the species' fitness (Bradshaw and Holzapfel 2010; Bedrosian et al. 2011). The introduction of artificial night lights can shift entire breeding phenologies in temperate zone birds, for example, given that singing behavior, timing of gonadal growth, and egg laying are all proximately controlled by photoperiod (Da Silva et al. 2015). In addition to birds, artificial lights have also been shown to influence the reproductive activities of bats, frogs, fish, and beetles (Table 3).

Continuous periods of darkness also play an important role in controlling the repair and recovery of certain physiological functions (Gaston et al. 2013). Darkness is vital to the production of melatonin, the hormone that orchestrates changes in body mass, metabolic rates, hormone synthesis, and immunity that, in turn, influence processes ranging from reproductive development to skin coloration to thermoregulation (Table 3; Zubidat et al. 2007; Da Silva et al. 2015; Dominoni et al. 2016). By disrupting the production of melatonin, artificial light can suppress species' immune responses, alter species' perception of day length, or change a species' metabolic rate requiring them to spend more time foraging (Leonardi and Klempau 2003; Navara and Nelson 2007; Perry et al. 2008; Da Silva et al. 2015). Constant illumination may even cause results as extreme as altering a species' gene expression (Perry et al. 2008).

MITIGATING THE IMPACTS OF ARTIFICIAL LIGHT

The impacts of artificial lighting on ecosystems can be mitigated using numerous approaches of varying intensity (Gaston et al. 2012). The most effective option would be to prohibit the use of artificial night lighting or restrict its use. Restrictions may include, for example, limiting the use of artificial lights to 1-2 hours following sunset and 1-2 hours preceding sunrise (vs. all night), switching lights off or dimming lights during critical times of the year such as when foraging, breeding, or dispersal and migratory activities are happening, or only allowing the use of motion-activated lights (Gaston et al. 2012; International Dark Sky Association- IDA 2019). Less restrictive options for mitigating the impacts of artificial night lighting are to ensure 1) lights are only used where they are needed, 2) lights are only illuminated when they are useful, 3) lights only illuminate the target area (i.e., trespass of light is minimized), and 4) lights are no brighter than necessary (IDA 2019). Trespass of light typically happens when lights are unshielded, which includes when light fixtures have an exposed bulb, and can be addressed by fully shielding fixtures and ensuring they are downward facing, as is required by CDFA for commercial cannabis cultivators (CDFA 2017; IDA 2019).

The impacts of artificial lighting may also be mitigated by changing the intensity or spectrum of the lighting (Gaston et al. 2012). Each type of lamp has a unique spectral signature, emitting light at differing intensities and over distinctive ranges of wavelengths (Gaston et al. 2013). This is true of both artificial light and natural light. In a natural photoperiod, for example, blue light increases as dusk falls, especially when the moon is new or absent (Sweeney et al. 2011). Blue light is then replaced by moonlight and/or starlight, which is red-shifted relative to sunlight (Sweeney et al. 2011). These spectral characteristics are used by wildlife species as sources of information regarding their location and the time of day, triggering numerous behavioral and physiological processes (Sweeney et al. 2011; Longcore et al. 2015). White light-emitting diodes (LEDs), which emit a large fraction of their energy as blue light, have rapidly become the most common type of outdoor lighting, with higher Color Correlated Temperature (CCT) LEDs emitting more blue light than lower CCT LEDs (e.g. a 4000° Kelvin CCT LED typically emits more than a 2700° Kelvin CCT LED). This may be problematic for local wildlife populations as blue light produces more sky glow than lower color temperatures (e.g., yellow or red light) and contains the most biologically active wavelengths for physiological processes like hormone production and daily activity (Gaston et al. 2012; Kyba and Hölker 2013; Brainard et al. 2015; IDA 2019).

The spectral composition of LEDs can be custom-built, however, to mitigate the effects of artificial night light on ecosystems (Table 4; Poot et al. 2008; Gaston et al. 2012; Ouyang et al. 2015; De Yong et al. 2018). The IDA (2019) recommends using LEDs with color temperatures less than 3000 Kelvins when white lighting is needed and there are no specific wildlife concerns. When there are wildlife concerns, the recommended spectral composition of LEDs is species-specific. Green, yellow, phosphor-coated amber, and white LEDs with filters that remove blue wavelengths have all been found to help minimize the responses of certain wildlife species to artificial light (Longcore and Rich 2016; Longcore et al. 2018).

FUTURE DIRECTIONS

Artificial lights associated with cannabis cultivation may differ from lights associated with other forms of human development both temporally and spatially. They may differ temporally if lights are on continuously during nighttime hours, as compared to motion-activated lights or lights that are only on in the daytime. They may differ spatially if lights are operating in areas that are predominantly rural and forested, as compared to lights that are clustered in housing developments or in large agricultural areas. As of August 2019, 43% of commercial cannabis cultivation licenses issued by CDFA have been for mixed-light cannabis cultivation, which uses artificial lights to extend the number of growing hours in a day and the number of growing days in a year (i.e., the lights function during nighttime hours). The majority of these mixed-light licenses have been issued in Humboldt and Mendocino counties in northwestern California, one of the least developed regions of the state, with most cannabis-related development in this region occurring in areas previously covered in natural vegetation, notably old growth and second growth forests (Butsic et al. 2018). While this suggests that artificial lighting associated with cannabis cultivation may be distributed differently across the landscape than other types of artificial lighting, empirical data are desperately needed. Thus, in relation to cannabis cultivation, we encourage assessments on 1) the proportion of cultivators using artificial light in an outdoor or mixed light setting, and whether these lights are fully contained (i.e., such that no light escapes), 2) the number of nighttime hours when artificial lights are illuminated and how this varies throughout the year, and 3) the spatial distribution of artificial light sources and resulting skyglow at both local (e.g., within a forested or urban environment) and statewide scales. This information is imperative for developing our understanding of how artificial lighting is used in cannabis cultivation, how it may be impacting fish and wildlife populations in California, and how we can proactively mitigate any potential impacts.

Table 4. Examples of how different light colors impact wildlife.

Color	Species	Response	Source
White	Nocturnally migrating birds	60.5 – 80.8% of observed birds were disoriented by and attracted to white light.	Poot et al. 2008
	Plecotus and *Myotis* bat species	These bat species avoided transects illuminated by white light (via light posts).	Spoelstra et al. 2017
	Pipistrellus bat species	These bat species were more abundant in transects illuminated by white light (via light posts) than in darkness, likely because of the accumulation of insects.	Spoelstra et al. 2017
	Common toads	Toads avoided sections of road illuminated in white light.	Grunsven et al. 2016
Red	Nocturnally migrating birds	53.8 – 54.2% of birds were disoriented by and attracted to red light.	Poot et al. 2008
	Common toads	The toads showed no response if the road was illuminated in red light.	Grunsven et al. 2016
	Plecotus, *Myotis*, and *Pipistrellus* bat species	Bats were equally abundant in transects illuminated by red light (via light posts) and in darkness, which suggests they were least disturbed by red light.	Spoelstra et al. 2017
	House flies	Flies were attracted to red light.	Longcore et al. 2015
Green	Nocturnally migrating birds	Birds were less disoriented by green light than by red and white light, with only 12.5 – 27.3% of observed birds reacting to green light.	Poot et al. 2008
	Plecotus and *Myotis* bat species	These bat species avoided transects illuminated by green light (via light posts).	Spoelstra et al. 2017
	Mosquitos, midges, house flies	These insects are attracted to green light.	Longcore et al. 2015
Blue	Nocturnally migrating birds	Birds were the least disoriented by blue light (2.7 – 5.3% of observed birds reacted), when compared to red, white, and green light.	Poot et al. 2008
	Most insects	Many insects are attracted to blue light.	Longcore et al. 2015
	Coho Salmon (*Oncorhynchus kisutch*) and Chinook Salmon (*Oncorhynchus tshawytscha*)	In a lab setting, salmonids were more active (90% increase in activity) under lights in the blue and ultraviolet spectrum (mercury vapor lamps), when compared to strobe lights.	Puckett and Anderson 1988, Nemeth and Anderson 1992

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Conceived and designed the study: LNR, EF, ADB
Collected the data: LNR, EF, ADB
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Authored the manuscript: LNR, EF, ADB, EC
Provided critical revision of the manuscript: EF, ADB, EC

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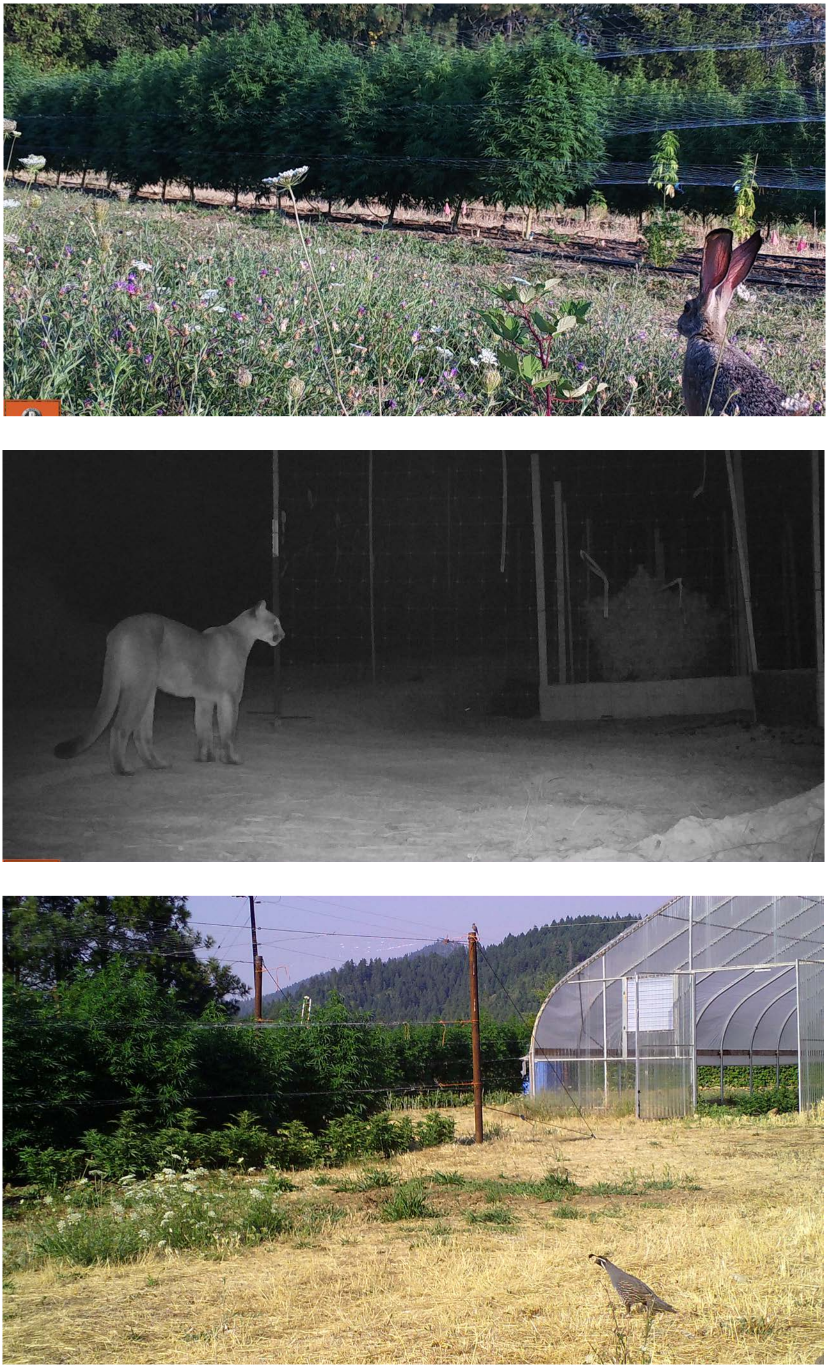

Examples of wildlife on cannabis cultivation sites in the Klamath-Siskiyou Ecoregion of southern Oregon [in order from top to bottom: black-tailed jackrabbit (*Lepus californicus*), mountain lion (*Puma concolor*), and California quail (*Callipepla californica*)]. Photo credit: Phoebe Parker-Shames, UC-Berkeley.

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Coexisting with cannabis: wildlife response to marijuana cultivation in the Klamath-Siskiyou Ecoregion

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The recent expansion of cannabis agriculture in rural areas of the western United States provides an ideal opportunity to study the outcomes of policy change for land use and wildlife. Small scale (<1 acre), private-land cannabis cultivation has the ability to coexist with or alter surrounding wildlife communities. To date, there has been little to no formal research on wildlife response to this form of cannabis cultivation. This study examines local wildlife community dynamics on and nearby active private-land cannabis farms. Using camera data collected between 2018–2019, we summarized the frequency of occurrence of 11 wildlife species and 3 domestic animals on and adjacent to (within 500 m) eight cannabis farms within the Klamath-Siskiyou Ecoregion in southern Oregon. We also assessed how cannabis production influenced the occupancy (defined here as space use) of black-tailed deer (*Odocoileus hemionus columbianus*), lagomorphs, and gray fox (*Urocyon cinereoargenteus*) in our study area. We found that cannabis farms were generally occupied by smaller-bodied wildlife species, and had a higher proportion of domestic dog (*Canis familiaris*), cat (*Felis catus*), and human activity compared to nearby comparison sites. The presence of a cannabis farm helped explain detection probabilities of deer and gray fox, but did not affect lagomorphs. Farms also helped predict gray fox occupancy, but were not selected in lagomorph or deer models. These results suggest species-specific responses to cannabis cultivation, and highlight the need for further research on site-level production practices and their influence on surrounding ecological communities.

Key words: agricultural frontier, anthropogenic disturbance, camera traps, cannabis cultivation, occupancy and detection, rural development, terrestrial mammals

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Cannabis agriculture is a quickly-expanding industry in the western United States, and represents a new opportunity to study the ecological outcomes of a policy change that has induced rapid rural land development (Carah et al. 2015; Butsic et al. 2018). Cannabis has been cultivated in the western United States for decades, but as a federally illegal drug it was confined to surreptitious farms, often in remote areas (Corva 2014; Butsic and Brenner 2016). Illegal production on public lands in the West have long elicited concerns about their potential environmental impact via habitat destruction, toxicant use, and poaching (Gabriel et al. 2012; Levy 2014; Carah et al. 2015). However, over the past few years, recreational legalization of cannabis in several states, including California and Oregon, has led to the expansion of production on private land, potentially supplanting much of the production on public lands (Arcview Market Research 2016; Butsic et al. 2018; Klassen and Anthony 2019). This expansion raises new questions about how the evolving cannabis industry may interface with and potentially alter surrounding wildlife communities and their habitats (Carah et al. 2015; Butsic et al. 2018).

While a robust body of literature on wildlife-agriculture interactions may help predict outcomes associated with the growth of cannabis agriculture, cannabis production has several unusual factors that differ from most other crops: 1) remoteness, 2) small individual farm size, and 3) unique spatial pattern on a landscape. Influenced by its illicit history, cannabis is often grown in remote, biodiverse regions with minimal other non-timber agriculture (Corva 2014; Butsic and Brenner 2016; Butsic et al. 2018). Regardless of individual legal status, private land cannabis farms are typically smaller than those of other commercial crops, and are clustered in space, creating a unique land use pattern of small points of development surrounded by less developed land (Butsic and Brenner 2016; Wang et al. 2017; Butsic et al. 2018). This pattern of development in rural areas, particularly in the West, makes cannabis agriculture a frontier industry—one that often occurs at the wilderness boundary—a somewhat rare characteristic for agriculture in the United States (Butsic et al. 2018).

Cannabis production practices vary widely depending on legality and land ownership, which influence the severity and type of environmental impacts from cultivation (Carah et al. 2015; Wilson et al. 2019). It is unclear how many of the environmental concerns associated with public land cultivation can be generalized to private land, but it is likely that many production practices differ. For example, reports and studies on illegal public land cannabis production list the following impacts from these sites: use of anticoagulant rodenticides and other toxicants that can bioaccumulate across food chains, poaching, habitat alteration, illegal water withdrawals, and potential water contamination (Gabriel et al. 2012; Levy 2014; Thompson et al. 2014; Carah et al. 2015; Gabriel et al. 2018). Yet, surveys of licensed and unlicensed cannabis producers in California suggest that toxicant use, poaching, and water contamination may be less prevalent with private land producers (Wilson et al. 2019). In addition, on many private land farms, both licensed and unlicensed, the use of high-powered grow lights, drying fans, and visual barrier fencing may create potential wildlife disturbance (Rich, Baker, et al this issue.; Rich, Ferguson, et al. this issue). Such practices are less common on public land. Given that regulated cannabis agriculture is an emerging industry that has prioritized sustainable cultivation, research on how private land cannabis farms may impact wildlife is conspicuously scant.

Here, we examine how private land cannabis cultivation may interface with wildlife communities on and surrounding outdoor cannabis farms (both licensed and unlicensed). Individual species may respond to different cues on cannabis farms (e.g., lighting, fences, human activity), and so we expected species would exhibit a range of responses including being deterred by, attracted to, or indifferent to cannabis production. For example, larger animals, such as black-tailed deer (*Odocoileus hemionus columbianus*), may be less likely to use cannabis farms due to fencing and increased human presence (Brashares 2010; Fischer et al. 2012). Alternatively, cannabis farms could attract species such as behaviorally flexible omnivores or foragers through the creation of novel food sources or new edge habitat (Gabriel and Wengert 2019). These individual responses may offer insights into broader community level responses.

The goal of this study was to understand local wildlife community composition on and nearby small, private cannabis farms. We used data from remotely triggered camera surveys (hereafter cameras) to determine whether cannabis cultivation influenced the local distributions of terrestrial mammals (>0.5 kg). To achieve this goal, our objectives were to use camera detections to: 1) describe species composition on and near cannabis farms, and 2) compare individual species habitat-use responses to cannabis production (using occupancy modeling where "occupancy" is redefined as "use" as in Latif et al. 2016). Ultimately, this research is intended to support efforts to predict and mitigate potential outcomes of cannabis development on terrestrial wildlife communities.

METHODS

Study area

Our study area was situated within the Oregon portion of the Klamath-Siskiyou Ecoregion and consisted of farms spread across three sub-watersheds (Slate Creek, Lower Deer Creek, and Lower East Fork Illinois River; defined by USGS hydrologic unit code 12) in Josephine County, southwestern Oregon (42.168, -123.647; Figure 1). We set cameras at 1,240 m to 1,910 m above sea level. The study area included a mix of vegetation types, including open pasture, serpentine meadows, oak woodland, and mixed conifer forest. Rainfall in this region varies seasonally and by elevation, with an average of 82.7 cm annually (Borine 1983). Mean temperatures ranged between 3.9-20.6°C in 2018–2019 (NOAA https://www.ncdc.noaa.gov/cdo-web/).

The Klamath-Siskiyou Ecoregion is one of the most biodiverse temperate forest regions on Earth, in an area that straddles the Oregon-California border and contains several regions identified as critical climate change refugia (Olson et al. 2006; Olson et al. 2012). Several species of concern are present in the county, including native salmonids, threatened Humboldt martens (*Martes caurina humboldtensis*), Pacific fishers (*Pekania pennanti*), and spotted owls (*Strix occidentalis*), all of which are hypothesized to be directly or indirectly affected by cannabis agriculture (Gabriel et al. 2012; Thompson et al. 2014; Carah et al. 2015; Gabriel et al. 2015; Butsic et al. 2018).

Southern Oregon, and Josephine County in particular, have a long history of illicit and medical cannabis cultivation, as well as an active presence in the growing legal industry in Oregon (Klassen and Anthony 2019; Smith et al. 2019). Southern Oregon has become known as a prime destination for outdoor cannabis production, and Josephine County has the highest number of licensed producers relative to population size in the state (Oregon Liquor Control Commission 2019; Smith et al. 2019). Production in the county accelerated after recreational legalization in 2014 (Parker-Shames, unpublished data), and takes a similar form to cultivation occurring across the border in northern California, with clusters of small farms surrounded by undeveloped or less developed rural land (Butsic and Brenner 2016; Wang et al. 2017; Butsic et al. 2018; Smith et al. 2019).

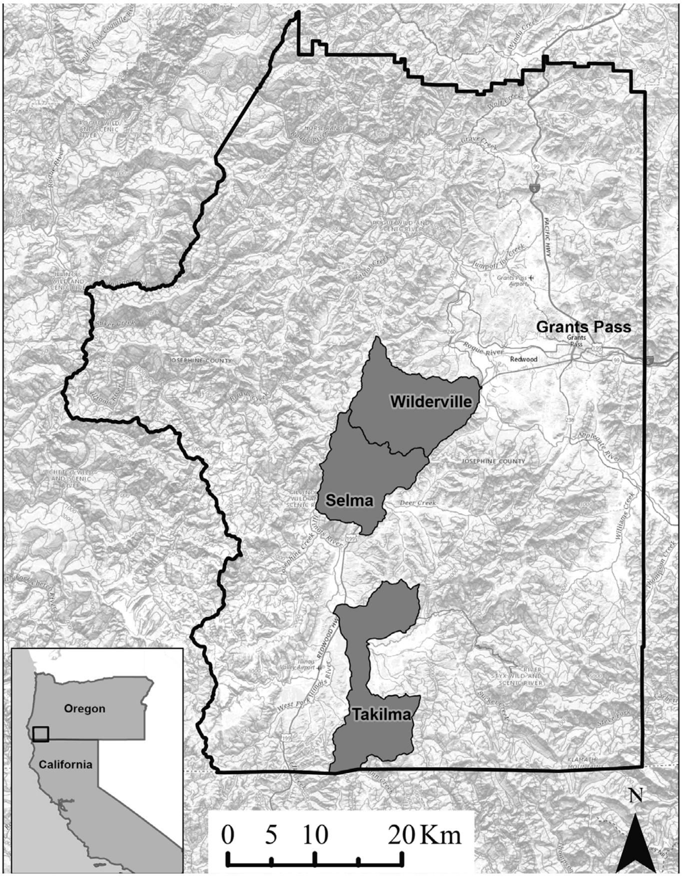

Figure 1. Map of study area with local population centers identified. The study sites are indicated as USGS hydrologic unit code 12 sub watersheds within Josephine County, southern Oregon. All study farms were contained within these three watersheds, and are summarized at this scale to anonymize specific farm locations. From the top down, the sub watersheds are: Slate Creek, Lower Deer Creek, and Lower East Fork Illinois River.

Cannabis farms for this study included one licensed recreational production site, one medically licensed (though non-compliant) production site, and six unlicensed sites. All farms were producing cannabis for sale, though in different markets depending on their access to licensed markets. We selected these eight farms because they (1) were representative of the size and style of cultivation predominant in Josephine County in the years immediately following recreational legalization in 2015 (Parker-Shames, unpublished data), (2) were all established after recreational legalization except for the medical farm, (3) did not replace other plant-based agriculture, and (4) granted us permission to set up cameras on site. Our sampled farms were small (typically < 1 acre), had conducted some form of clearing for production space, and three had constructed some form of fence or barrier around their crop. Nonetheless, specific land use practices and production philosophies differed between farms (e.g., pesticide use, type of fencing, presence of dogs, number of people working on the site, attitudes towards conservation, etc.). We cannot disclose farm locations, as per our research agreement for access.

Camera surveys

Monitored farms were clustered within each watershed: one farm in Slate Creek, five in Lower Deer Creek, and two in Lower East Fork Illinois River. We placed un-baited motion sensitive cameras (Bushnell E3, Bushnell Aggressor, or Moultriecam models) on and surrounding cannabis farm clusters as well as in random locations up to 1.5 km from the farms. To guide the placement of cameras, we overlaid the area surrounding each cannabis farm cluster with a 50 x 50 m grid and then selected a random sample of at least one-quarter of grid cells (a minimum of 45 locations in each watershed), stratified by vegetation openness and distance to cannabis farm. We rotated 15-20 cameras through the sampled grid cells, ensuring each camera was deployed for a minimum of two weeks. As a result of sampling across two years, we likely violated the model's assumption of geographic and demographic closure (Mackenzie et al. 2006), but given our interest was in space use associations and not estimates of occupancy, we believe this is a minimal issue. For this analysis, we restricted our data to a subset of cameras on cannabis farms ("cannabis sites") and cameras in 500 m proximity to farms ("comparison sites") active during the same camera rotation (n = 8 farms, 17 rotations, 2-5 cameras/rotation). Because of rotations and field constraints, all cannabis sites were not monitored at the same time or for the same length of time (one to six rounds). Each cannabis site had at least one, and up to three comparison cameras within 500 m during each of its active rounds. Because of farm clustering, some comparison cameras were within 500 m of more than one farm. Half the cameras on farms (n = 4) were monitored for more than one round, but the comparison camera(s) were not always the same for all rounds due to rotations.

Statistical analyses

We summarized species observations at cannabis farms and created detection histories (i.e., matrices where a "1" indicated the species was photographed at a given camera station during the respective 24-hr time interval, a "0" that it was not, and an NA if the camera was inactive) using the package CamtrapR (CamtrapR v.1.2.3, https://cran.r-project.org/web/packages/camtrapR/index.html, accessed 11 December 2019) in program R (R v.3.6.2, www.r-project.org, accessed 18 December 2019). We used a 24-hr time interval because our focus was on estimating space use associations instead of occupancy, and a short interval reduced the likelihood of the same individual animal being detected on both the farm and comparison camera (Latif et al. 2016; Steenweg et al. 2018). We used the detection matrix to summarize detection rates per 100 operation nights for species found on cannabis sites and comparison sites. We then modeled the occupancy probabilities of the three most commonly detected wild species, which included black-tailed deer, lagomorphs (including brush rabbit *Sylvilagus bachmani* and black-tailed jackrabbit *Lepus californicus*), and common gray foxes (*Urocyon cinereoargenteus*), using the UNMARKED package in Program R (unmarked v.0.13-1, https://cran.r-project.org/web/packages/unmarked/index.html, accessed 11 December 2019). We combined lagomorphs due to uncertainties in distinguishing individual species in photographs.

We used single-species occupancy models to assess factors influencing the likelihood that a species used the area around each camera station (interpreting the "occupancy" parameter as "use" in that it is influenced by both occupancy and availability) and the probability that the species would be detected given they were present (i.e., “detection probability”, as influenced by both availability and perceptibility) (Latif et al. 2016). In this case, detection can also be influenced by fine scale activity and/or habitat use patterns (Latif et al. 2016; Moreira-Arce et al. 2016)

We hypothesized that cannabis cultivation, elevation, water access, and vegetation type would influence species’ spatial relationships, and therefore included them as predictors of occupancy (i.e., space use) in the model. We predicted that cannabis cultivation would have a negative influence on a species’ probability of using an area. We included a binary, categorical variable in the models to characterize whether detection occurred on a cannabis site (1) or a nearby comparison site (0). This variable reflected and distilled the on-site practices that are common across farms, including increased human activity and fencing. We expected regional elevation to influence species’ vegetation use, and therefore used the average elevation within a 1 km buffer of each camera location, from the 30 m National Elevation Dataset (State of Oregon Geospatial Enterprise Office, U.S. Geological Survey, www.gis.oregon.gov). Water access is frequently an important predictor for wildlife occupancy (Rich et al. 2019), especially during dry periods such as during our study years, so we included distance to streams as a predictor of occupancy (NOAA Intrinsic Potential Streams, https://archive.fisheries.noaa.gov/wcr/maps_data/maps_and_gis_data.html). To represent vegetation, we used the percent evergreen forest, as determined via the National Land Cover Database (NLCD 2016, www.mrlc.gov) within a 1 km buffer of each camera site as a vegetation predictor variable. Finally, to distinguish general biogeographic variation between regions, we used watershed as a categorical predictor for occupancy (we assigned cameras as Slate Creek = 1, Lower Deer Creek = 0, and Lower East Fork Illinois River = -1).

For modeling detection, we hypothesized that cannabis production sites would negatively influence the probability that a species was photographed given they were available in the general area, due to both physical barriers to wildlife accessing these sites, and to behavioral shifts, such as animals moving less or moving more cautiously around areas of higher human activity (Smith et al. 2017; Jakes et al. 2018; Tucker et al. 2018). We used distance to road (Josephine County GIS Department 2018) as a proxy for human activity separate from cannabis production that might also negatively influence detection probability. Although cannabis cultivation can be associated with the creation of new roads (Carah et al. 2015), the roads used in these analyses were not those created or used exclusively by cultivators. Finally, we included year as a categorical variable to account for potential inter-annual variation in detection ability.

We standardized covariates to have a mean of zero and a standard deviation of one. We used Akaike’s Information Criterion (AIC) (Burnham and Anderson 2002) to compare model fits. We modeled all of the detection covariates first, and then kept our top ranked model for detection constant before modeling our occupancy (use) covariates. We used our top ranked model to assess covariate relationships and determine which variables influenced species use and probabilities of being photographed.

RESULTS

We analyzed over 5,000 animal detections over 957 operation nights (with an average of 31 operation nights per camera). We found that the communities of wildlife present on cannabis farms were qualitatively different from the surrounding, uncultivated areas

(Figure 2). Wildlife on cannabis farms were often smaller-bodied species, and co-occurred with higher human and domestic dog (*Canis familiaris*) activity. There were 18 different species recorded on cannabis farms, and 24 on comparison cameras. Wild predators were predominantly detected on comparison cameras rather than cannabis farms. For example, gray foxes had 18.5 detections per 100 operation nights on cannabis sites compared to a detection rate of 31.6 on comparison sites, while black bears (*Ursus americanus*) had a detection rate of 2.5 on cannabis sites compared to 4.9 on comparison sites, and coyotes (*Canis latrans*) had a rate of 1.9 on cannabis sites and 6.1 on comparison sites. By contrast, domestic predators such as cats (*Felis catus*) and dogs, had a detection rate twice as high on cannabis production sites than comparison sites (Figure 2). It is also worth noting detections of two rarer carnivores: we detected mountain lions (*Puma concolor*) seven times on a cannabis farm and once on a comparison site, and bobcats (*Lynx rufus*) two times on each.

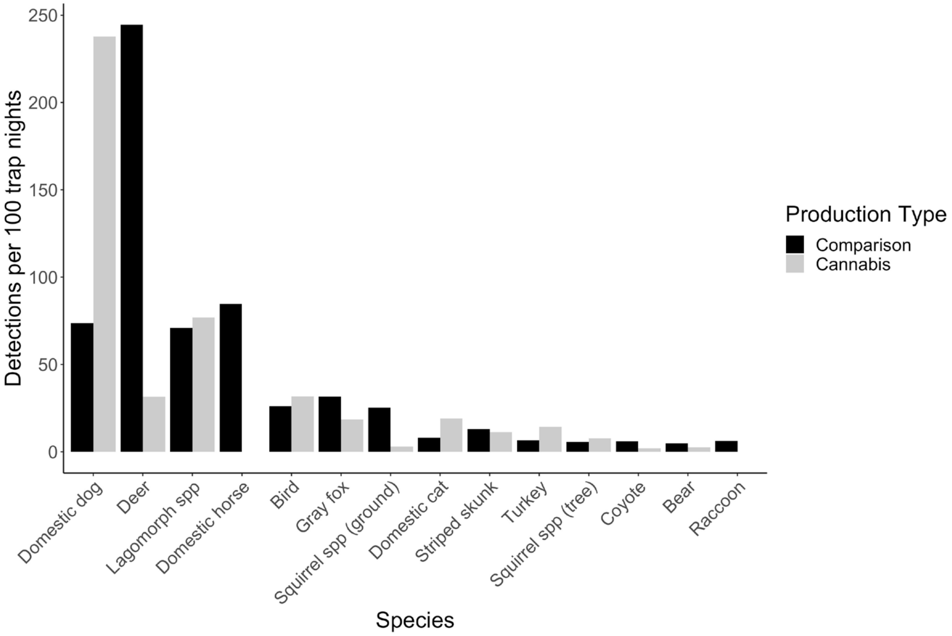

Figure 2. Top animal species present at cannabis (8) versus comparison (24) sites, by detection rate (number of detections per 100 sampling occasions). Excludes any species detected less than a total of 10 times at all sites combined. Excludes humans, which were the most frequently detected presence on both cannabis and comparison sites (detection rate of 1306.6 on cannabis sites and 478.7 on comparison sites).

For the single species occupancy models, detection variables varied by species. The top models for deer and gray foxes included a negative association with cannabis production for detection, while the top model for lagomorphs did not have similar associations (Table 1). Distance to roads was retained in all models for detection, and was positively associated with detection for all species, such that detection increased with increasing distance from roads.

For occupancy, here defined as use, cannabis production had a weak negative association with gray fox occupancy, and was not a top occupancy variable for any of the other species (Table 1). Because watershed and forest cover were correlated ($R^2 = 0.86$), we only used the variable with the highest univariate effect size for each species. For instance, watershed had a higher univariate effect size than forest cover for deer and gray fox occupancy, so we used watershed for candidate selection in those models, and forest cover for lagomorphs. No single variable was consistently selected as a predictor of occupancy across all species.

Table 1. Results from the top space use models for each species, including occupancy (defined in this case as use) and detection (influenced by both availability and perceptibility) variable beta estimates and 95% confidence intervals in parentheses. Stars are on confidence intervals that don't overlap zero.

	Occupancy Variables				
Species	Cannabis Production	Watershed	Elevation	Forest within 1 km	Distance to Streams
Black-tailed deer		-2.82 (-5.37– -0.27)*			
Gray fox	11.17 (-102.4–124.7)		-1.18 (-2.61–0.25)		
Lagomorphs				-0.99 (-2.29–0.30)	-0.73 (-2.32–0.86)
	Detection Variables				
Species	Cannabis Production	Year 2018	Year 2019	Distance to Roads	
Black-tailed deer	-1.71 (-2.26– -1.16)*	-0.485 (-1.02–0.05)	0.519 (-0.01–1.05)	0.522 (0.30–0.74)*	
Gray fox	-1.81 (-2.33– -1.29)*			1.81 (1.21–2.41)*	
Lagomorphs		0.45 (-0.021–0.92)	4.25 (2.85–5.66)*	0.77 (0.45–1.09)*	

DISCUSSION

This study represents a first step to quantify patterns of wildlife avoidance and coexistence on and surrounding active small-scale cannabis farms on private land. Our observational monitoring data suggest that wildlife species may be affected by these locations and may be altering their use of these environments. Specifically, our results suggest that 1) wildlife are consistently present on and around cannabis farms, 2) private land cannabis production may influence the local space use of some species more than others, and 3) cannabis farms may deter larger-bodied wildlife species in particular. Although limited by a small dataset, these results offer valuable insights into the ecological outcomes of the emerging cannabis industry.

The assessment of wildlife detection rates suggest that many wildlife species are consistently present at cannabis production sites (Figure 2, Figure 3). Whereas some species detected on cannabis farms are ones that have been recorded in the western United States as more tolerant to agriculture or disturbance (e.g. striped skunk, raccoon, deer), others are species that tend to avoid human activity (e.g. mountain lion, bobcat) (Crooks 2002; Gehring and Swihart 2003; Hilty and Merenlender 2004; VerCauteren et al. 2006). While we did detect some relatively rare species (mountain lion, bobcat), we did not detect others such as fishers or ringtails (*Bassariscus astutus*), and cannot assess whether this is due

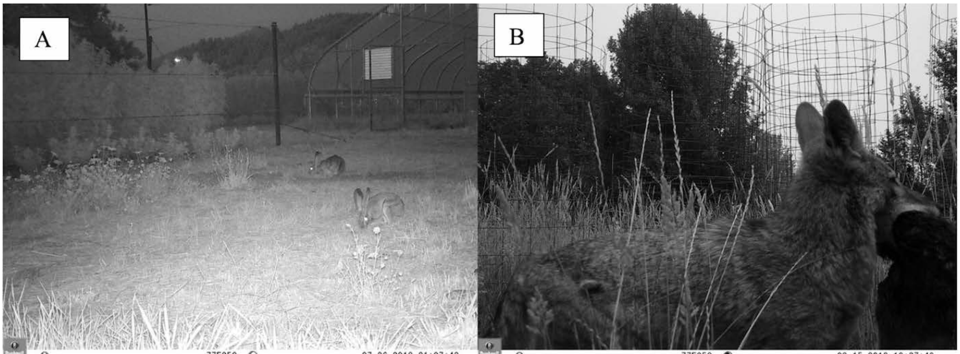

Figure 3. Examples of photos from cameras at cannabis sites demonstrating varied space use by wildlife at cannabis farms. A) Two black tailed jackrabbits (*Lepus californicus*) resting and foraging. B) A coyote (*Canis latrans*) passing through a cannabis farm with a prey item in its mouth. C) A squirrel (*Sciurus* sp.) passing in front of the camera carrying an unknown food item. D) A black-tailed deer (*Odocoileus hemionus columbianus*) appearing to forage at the base of a cannabis planter (the wooden box holding the cannabis plant in the photo).

to true absence or simply short study duration. We infer detection of wildlife on cannabis farms implies a potential for these species to move through these areas. In addition, some photos revealed foraging or resting behavior (Figure 3), which may indicate that cannabis agriculture could maintain biodiversity as other small scale agricultural crops have in other systems (Mendenhall et al. 2016). However, understanding long term impacts of cannabis production would require information on farm-level land use practices. For example, if animals on private land cannabis farms suffer fitness consequences similar to the toxicant exposure occurring on public land production, then coexistence on these sites may be detrimental in the long term (Levy 2014; Thompson et al. 2014; Carah et al. 2015; Gabriel et al. 2015; Gabriel et al. 2018).

Modeled use and detection probability results indicate that despite a general wildlife presence at cannabis farms, some animals may be more affected by these areas than others. For detection, both deer and gray fox were influenced by cannabis farms (Table 1). Distance to roads was positively associated with all species detection, suggesting that animals are consistently avoiding roads, but no other variable was consistent across all species for either detection or use. For occupancy (i.e. space use), cannabis farms were not selected for deer or lagomorph models (Table 1), but we suspect this could have been due to our close proximity of cannabis and comparison locations. It is possible that these species would move >500m within a 24-hour period, making it difficult to distinguish space use. Additionally, because we pooled lagomorph species, it is possible that either brush rabbits or black tailed jackrabbits individually might have responded differently to cannabis production. Nonetheless, cannabis farms influencing detection probabilities for deer and gray foxes may imply an influence on repeated visits over our time period, and potentially a behavioral adjustment near cannabis farms.

There are many possible explanations for why deer and gray fox space use might be more influenced by cannabis farms than lagomorphs. These generally have to do with characteristics on the farms themselves. Wildlife may be interacting with the increased presence of domestic cats and dogs on cannabis farms (Figure 2), for instance, for deer as potential or perceived prey, or gray foxes as competitors (Zapata-Ríos and Branch 2016; Reilly et al. 2017; Twardek et al. 2017). Alternatively, deer and gray foxes may be responding to behavioral cues from increased human presence and activity on cannabis farms (Berger 2007; Tucker et al. 2018). Lagomorphs may be responding to these same cues, but via different response mechanisms. It is possible that lagomorphs are more behaviorally flexible than deer and gray foxes and can avoid altering their spatial patterns by instead shifting their temporal activity patterns, for instance, becoming more nocturnal (Gaynor et al. 2018). More research is needed to disentangle these potential mechanisms.

Both detection rate summaries and model results suggest that cannabis farms appeared to disproportionately influence the space use of larger wildlife species. Black bears had a higher detection rate on comparison sites compared to cannabis farms (Figure 2) and the model results indicate that larger black-tailed deer and gray foxes might avoid cannabis farms, while smaller animals such as lagomorphs appear to be unaffected. This result is expected, as large bodied animals such as deer may be unable to access space on the farms if they are physically blocked by fencing, while smaller species are still able to move through these barriers (Brashares 2010; Jakes et al. 2018).

Despite variation in which species responded to cannabis farms, we did not find evidence from either detection rate summaries or model results to suggest that predators were attracted to these sites. Other studies have shown predators tend to avoid agricultural development, and our results seem to support the same trend (Gehring and Swihart 2003; Hilty and Merenlender 2004). By contrast, there has been recent suggestion that cannabis production on public lands may serve as an "ecological trap" by attracting carnivores to production areas that then expose individuals to deadly toxicants (Gabriel and Wengert 2019). Our results, at least in the short-term, suggest that this dynamic may be less likely to occur on small-scale private land cannabis farms. This highlights the different potential ecological threats and processes playing out on public versus private land cannabis production sites. Not only do private land cannabis farms seem to use fewer toxicants (Wilson et al. 2019), but they may also have higher human activity levels on site compared to public land production located in more remote areas. Wildlife may in turn tend to avoid this human presence rather than being attracted (Smith et al. 2017).

This study begins the discussion regarding a glaring shortage of data on animal space use on cannabis sites, but there are many further avenues for future research. For example, the relative importance of cannabis farms in their influence on animal space use should be analyzed in the surrounding landscape context. The influence of roads on the modeled detection results implies that cannabis cultivation, despite occurring in a rural area in this case, was not the only form of human disturbance to which animals were responding. It may therefore be useful to compare cannabis and other forms of rural land use. In addition, it is necessary to conduct further study at broader spatial and temporal scales in order to examine long term wildlife community response to cannabis and unravel the complicated set of potential contributing factors.

Management and conservation implications

Wildlife are likely to have species-specific responses to small-scale outdoor cannabis farms, and, thus, the specific land use practices occurring at a site may be influential for biodiversity conservation in these communities. Future studies should examine the role of fencing, timing of human activity, presence of domestic dogs and cats, and other site level practices that may influence wildlife use. Many small-scale cannabis farms are not part of a licensed production system (such as most included in this study), and therefore cannot be regulated for their production practices (Polson 2015; Short Gianotti et al. 2017). For these producers, a mix of educational resources on wildlife friendly growing practices, grower-enforced community standards or expectations, and law enforcement efforts to reduce noncompliance, may play an important role in increasing or maintaining biodiversity. For species deterred from cannabis farms, such as was implied by our deer and gray fox results, further research is needed to understand the mechanism for this avoidance. If, for example, fencing, artificial lighting, or sound are identified as major causes of this deterrence, then careful consideration should be given to the regulations on these practices at cannabis farms and their relation to critical habitat features such as water sources or animal migration routes.

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Author Contributions

Conceived and designed the study: PPS, advised by JB, and with consultation from LR
Collected the data: PPS
Performed the analysis of the data: PPS and WX, with guidance from LR
Authored the manuscript: PPS
Provided critical revision of the manuscript: PPS, WX, LR, and JB

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Generators such as this are common at cannabis cultivation sites, particularly those in remote forested regions like the Emerald Triangle (Humboldt, Mendocino, and Trinity Counties), CA. Photo Credit: CDFW staff

Example of a large outdoor cultivation in Humboldt County. Photo Credit: Scott Bauer, CDFW

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Anthropogenic noise: potential influences on wildlife and applications to cannabis cultivation

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Biological sounds play an important role in activities ranging from territory defense to mate choice to predator avoidance to foraging. Anthropogenic noise can mask these sounds, potentially altering the habitat selection, activity patterns, phenology, and physiology of wildlife species. For example, cannabis (Cannabis sativa or C. indica) cultivation may increase levels of anthropogenic noise given the use of diesel generators, irrigation pumps, and landscaping equipment. To predict how noises associated with cannabis cultivation may influence wildlife in California, we review scientific literature assessing the influences of anthropogenic noise on various species of mammals, birds, herpetofauna, and invertebrates. We then outline potential noises associated with cannabis cultivation and why they may be unique on the landscape and provide recommendations on future research needs.

Key words: activity patterns, anthropogenic noise, cannabis, habitat selection, phenology, physiology, wildlife

The acoustic environment is more than just a collection of auditory signals between individuals, it is an interconnected landscape of information networks consisting of many signalers, receivers, and sounds vital to the fitness of a species (Templeton and Greene 2007; Barber et al. 2010; Read et al. 2013). For example, sounds pertaining to territory defense, mate attraction, or family cohesion (i.e., contact calls) promote reproductive success (Halfwerk et al. 2011a, b; Allen et al. 2016). In songbirds, these sounds are used to assess numerous individuals simultaneously for mate choice, extra-pair copulations, and rival assessment (Barber et al. 2010). Alternatively, sounds announcing the approach of predators (i.e., alarm calls) promote survival of both conspecifics to whom the calls were directed and other species that capitalize on the alarms (Templeton and Greene 2007; Sloan and Hare 2008; Magrath et al. 2015).

Successful acoustic communication requires sounds to 1) move through the environment from senders to receivers and 2) be detectable through background noise (Patricelli and Blickley 2006). There is mounting evidence that noise produced by humans, whether from vehicles, construction equipment, or humming power sources (e.g., generators, power lines, wind turbines), dramatically increases the amount of background noise, in turn impeding detectability of acoustic signals and negatively impacting the ability of a species to communicate (Fernández-Juricic et al. 2005; Gillam and McCracken 2007; Barber et al. 2010; Kite and Swaddle 2011; Francis and Barber 2013). Masking of biologically relevant sounds can limit mate choice, cause species to abandon territories or potential habitat, negatively impact species' ability to locate food, or cause deleterious physiological effects like hearing loss, raised blood pressure, and increased production of stress hormones (Rabin et al. 2006; Wright et al. 2007; Schaub et al. 2008; Shannon et al. 2014; Ware et al. 2015). In a rural to suburban area where ambient noise levels are 45 – 55 decibels (dB), new sources of anthropogenic noise can begin having deleterious effects when they increase overall noise by just 5 – 10 dB (Dooling and Popper 2007). The specific noise level at which impacts begin to appear, however, depends on the amount of ambient noise and the temporal and spectral overlap between anthropogenic and biological sounds (Dooling and Popper 2007; Halfwerk et al. 2011). Species with low-frequency vocalizations like owls and grouse tend to have the largest spectral overlap with traffic noise, for example, which means these species are more likely to have their mate attraction or territorial defense songs obscured by human-produced noises (i.e., experience a decline in signaling efficiency; Slabbekoorn and Ripmeester 2007; Bunkley et al. 2015).

Cannabis cultivation has the potential to add additional sources of anthropogenic noise into a landscape through, for example, diesel generators, irrigation pumps, climate control systems, landscaping equipment, and vehicles. There is concern that this additional anthropogenic noise may reach the level of take, as defined by the Federal Endangered Species Act (ESA; i.e., an action of or attempt to hunt, harm, harass, pursue, shoot, wound, capture, kill, trap, or collect a species), for sensitive species like the northern spotted owl (*Strix occidentalis occidentalis*) and marbled murrelet (*Brachyramphus marmoratus*; USFWS 2006). For northern spotted owl and marbled murrelet it was determined that disturbance may reach the level of take if 1) project-generated sound exceeds ambient nesting conditions by 20-25 dB, 2) project-generated sound, when added to existing ambient conditions, exceeds 90 dB, or 3) human activities occur within a visual line-of-sight distance of 40 m or less from a nest (USFWS 2006). We note that California's ESA has a narrower definition of take (i.e., any action of or attempt to hunt, pursue, catch, capture, or kill). This could make it more difficult to directly attribute take to anthropogenic noise under the California ESA when compared to the Federal ESA.

Information on the levels of noise produced by cannabis cultivation specifically and the subsequent influences on wildlife species, however, is scant. To predict how anthropogenic noise associated with cannabis cultivation may influence wildlife in California, we reviewed scientific literature that assessed the influences of human-produced noise on species' habitat selection, activity patterns, phenology, and physiology. We then provide recommendations on future research needs.

Habitat selection and Activity Patterns

Mobile animals are often guided by sound, with conspecific signals attracting group members or potential mates, heterospecific signals (i.e., signals from a different species) indicating suitable habitat, and overall soundscape signals providing cues for general orientation (Slabbekoorn and Bouton 2008). Consequently, site abandonment and changes in habitat selection and activity patterns are among the most detected impacts of noise (Table 1; Francis and Barber 2013). Species ranging from deer to songbirds to frogs have been documented avoiding areas with anthropogenic noise, in turn influencing both fine-scale habitat selection and large-scale patterns of movement (Table 1; Sawyer et al. 2006; Mukhin et al. 2008; Francis et al. 2011; Ware et al. 2015; Caorsi et al. 2017). Further, avoidance or use by one species may lead to avoidance or use by others. This has been documented in nocturnally migrating bird species, where migrant birds listen for the heterospecific calls of resident birds to make decisions about which habitats to use as stopover sites (i.e., the heterospecific attraction hypothesis; Mönkkönen et al. 1990; Mukhin et al. 2008). It has

Table 1. Examples of changes in habitat selection and activity patterns resulting from anthropogenic noise.

Taxa	Species	Response	Source
Mammals	Mule deer (*Odocoileus hemionus*)	Radio-collared deer were more likely to occupy habitat away from noise-producing oil and gas developments than habitat in close proximity; changes in habitat selection happened within 1 year of development and there were no signs of acclimation.	Sawyer et al. 2006
	Sonoran pronghorn (*antilocapra Americana sonoriensis*)	Pronghorn at a military site where there was noise from overflights, ordinance deliveries, and human activity foraged less and stood and traveled more than pronghorn not exposed to military activity.	Krausman et al. 2004
	California ground squirrels (*Otospermophilus beecheyi*)	Close to wind turbines, where noise levels were higher than control sites (110.2 dB vs. 79.8 dB), squirrels exhibited increased rates of vigilance and were more likely to return to their burrows during alarm calling (i.e., increased caution).	Rabin et al. 2006
	Prairie dogs (*Cynomys ludovicianus*)	When exposed to road playback noise (77 dB at 10m), the number of prairie dogs aboveground decreased by 21%, the proportion of individuals foraging decreased by 18%, and vigilance increased by 48%. These results were consistent across a 3-month period suggesting there was no habituation.	Shannon et al. 2014
	Bat community	Bat species emitting low frequency (< 35 kHz) echolocation calls had a 70% reduction in activity levels at loud compressor sites (70 – 82 dB) vs. quieter well pads (53 – 70 dB). Bat species emitting high frequency calls did not show altered activity levels.	Bunkley et al. 2015
	Greater mouse-eared bat (*Myotis myotis*)	Successful foraging bouts decreased, and search time increased with proximity to acoustically simulated highway noise. At 7.5m from the noise source, it took the bats 5x longer to find their prey, which they locate by listening for faint rustling sounds.	Siemers and Schaub 2011
Birds	American robin (*Turdus migratorius*)	Foraging success was reduced when the auditory cues that robins rely on to locate buried worms were obscured by white noise (61 dB).	Montgomerie and Weatherhead 1997
	Nocturnally migrating birds	To test the effect of noise alone, a "phantom road" was created through an array of speakers broadcasting traffic noise. Among the bird community, 31% avoided using the phantom road as a stopover site during migration and the birds that did use the site showed a decrease in their overall body condition.	Ware et al. 2015
	Grey flycatcher (*Empidonax wrightii*)	Occupancy of flycatchers was lower at sites with 46-68 dB of noise than sites with 32-46 dB of noise.	Francis et al. 2011

Table 1. continued.

Taxa	Species	Response	Source
	White-throated sparrow (*Zonotrichia albicollis*), yellow-rumped warbler (*Dendroica coronata*), and red-eyed vireo (*Vireo olivaceus*)	Passerine density was 1.5x higher at energy sites that did not produce noise than at those that did (48 dB).	Bayne et al. 2008
	Greater sage-grouse (*Centrocercus urophasianus*)	Radio-marked female grouse were more likely to select habitat away from noise-producing oil and gas developments and were 1.3x more likely to occupy sagebrush habitats lacking wells within a 4-km^2 area.	Doherty et al. 2008
Herpetofauna	Bischoff's tree frog (*Boana bischoffi*) and fine-lined tree frog (*B. leptolineata*)	Both species moved away from playbacks of road noise (played at two intensities- 65 and 75 dB), suggesting the noise resulted in their spatial displacement.	Caorsi et al. 2017

also been documented in marbled newts (*Triturus marmoratus*) and smoot newts (*Lissotriton vulgaris*), which orient towards the calls of species that share similar breeding habitat (Diego-Rasilla and Luengo 2004; Pupin et al. 2007). _

Sound is also important in determining how much time and energy a species expends on activities like resting, vigilance, and foraging (Quinn et al. 2006; Rabin et al. 2006; Shannon et al. 2014). Many animals use sound to detect approaching predators or to warn conspecific and heterospecific co-occurring species (e.g., through alarm calls) that a predator is approaching. Quiet environments facilitate detection of these auditory cues, so less time needs to be spent searching for predators. Conversely, noisy environments impede auditory cues resulting in species spending more time and energy on anti-predator behaviors like vigilance and caution (e.g., not traveling far from a burrow; Quinn et al. 2006; Shannon et al. 2014). A positive relationship between noise and predator avoidance has been documented in both mammal and bird species (Quinn et al. 2006; Francis and Barber 2013; Shannon et al. 2014). California ground squirrels (*Otospermophilus beecheyi*), for example, tend to exhibit increased rates of vigilance in noisy environments where their ability to hear conspecific alarm calls is hindered (Rabin et al. 2006). If noise causes ground squirrels to miss just a single conspecific alarm call, then they may underestimate potential threats and in turn, increase their exposure to predation (Sloan and Hare 2008). In chaffinches (*Fringilla coelebs*) and prairie dogs (*Cynomys ludovicianus*), alternatively, noise leads to more time expended on vigilance and less time on foraging (Quinn et al. 2006; Shannon et al. 2014). Delayed response times of ground squirrels and loss of foraging time in chaffinches and prairie dogs demonstrate how noise, through its influence on predator-prey dynamics, can have both immediate (i.e., survival) and long-term (i.e., decreased nutrition/energy) impacts on species' fitness (Frid and Dill 2002).

Lastly, anthropogenic noise may decrease foraging efficiency if the species relies on auditory cues to locate food. Bat species specialized in gleaning arthropods off vegetation or the ground, for example, find prey by passively listening for prey-produced sounds

(Schaub et al. 2008). Thus, in environments with more noise, gleaning bats have fewer successful foraging bouts and spend more time searching for prey (Table 1; Schaub et al. 2008; Siemers and Schaub 2011). Decline of 12 species of bats in California that are either endangered or species of special concern has been correlated to reduced foraging success in noisy environments (Schaub et al. 2008; Siemers and Schaub 2011). Bird species like American robins (*Turdus migratorius*), marsh hawks (*Circus cyaneius*), and barn owls (*Tyto alba*), as well as reptile species like geckos (*Hemidactylus tursicus*), also use auditory cues to detect and locate prey. Like gleaning bats, these species have reduced foraging success in noisy environments where cues are obscured (Knudsen and Konishi 1979; Rice 1982; Sakaluk and Belwood 1984; Montgomerie and Weatherhead 1997).

Phenology and Physiology

To mitigate the negative impacts that anthropogenic noise may have on acoustic communication, many species adjust the frequency structure (i.e., pitch), amplitude (i.e., loudness), or timing of their vocalizations (Table 2; Patricelli and Blickley 2006). Vocal adjustments have been documented in a range of species, including bats, birds, frogs, and insects (Table 2). Brazilian free-tailed bats (*Tadarida brasiliensis*), reed buntings (*Emberiza schoeniclus*), great tits (*Parus major*), cicadas (*Cryptotympana takasagona*), and grasshoppers (*Chorthippus biguttulus*), for example, use higher call frequencies in the presence of anthropogenic noise (Slabbekoorn and Peet 2003; Gillam and McCracken 2007; Gross et al. 2010; Lampe et al. 2012; Shieh et al. 2012). Conversely, various species of frogs often increase or decrease their call rates based on the level of background noise (Lengange 2008; Cunnington and Fahrig 2010; Vargas-Salinas and Amézquita 2013). The benefit of vocal plasticity is that it allows species to adjust to new, noisy conditions (Gross et al. 2010). The hindrance is that it may negatively impact species' fitness by reducing transmission distances (e.g., high frequency signals attenuate faster), increasing the risk of predation or parasitism by making animals more conspicuous, altering energy budgets causing vital information to be lost (e.g., for mate choice), or breaking down signaler-receiver coordination (Luther 2008; Read et al. 2013).

In addition to altering the phenology of a species, exposure to noise can also influence the physiology of a species. Ungulates, bears, whales, game birds, songbirds, and frogs have all been documented to have adverse physiological responses to anthropogenic noise (Table 2; Powell et al. 2006; Rolland et al. 2012; Troianowski et al. 2017). These responses include hearing loss, hypertension (i.e., raised blood pressure), and increased production of glucocorticoids or stress hormones (Wright et al. 2007; Dooling and Popper 2007; Shannon et al. 2016). Increased production of stress hormones can in turn, negatively impact the survival and reproduction of a species by causing decreased immune response, diabetes, or reproductive malfunctions (Kight and Swaddle 2011; Tennessen et al. 2014). Exposure to noise led to increased stress hormone levels in European tree frogs (*Hyla arborea*), for example, which led to an immunosuppressive effect (Troianowski et al. 2017). The severity of a species' physiological responses is likely dependent on season. Northern spotted owl (*Strix occidentalis caurina*) males, for example, had the strongest response to motorcycle noise in May, when feeding themselves, their mates, and their nestlings (Hayward et al. 2011). The physiological response of migratory birds, alternatively, may be most acute mid-migration when maintenance of body condition is particularly imperative (Ware et al. 2015).

Table 2. Examples of phenological and physiological changes associated with anthropogenic noise.

Taxa	Species	Response	Source
Mammals	General	If the inner ear sensory hair cells are damaged, then mammals will experience permanent hearing loss.	Dooling and Popper 2007
	Brazilian free-tailed bats (*Tadarida brasiliensis*)	Bats recorded in the presence of high-frequency sounds used higher call frequencies than bats recorded in silence, which suggests that bats adjusted their echolocation call structure to minimize acoustic interference.	Gillam and McCracken 2007
	Desert mule deer (*Odocoileus hemionus crooki*) and desert bighorn sheep (*Ovis canadensis mexicana*)	Heart rates of captive animals increased relative to dB levels (from simulated jet aircraft noise) but returned to pre-disturbance levels within 60-180 seconds.	Weisenberger et al. 1996
Birds	House finches (*Carpodacus mexicanus*)	Males increased the low frequency (1.62 kHz) of their songs in areas with higher ambient noise to reduce the masking effects of the noise.	Fernández-Juricic et al. 2005
	Ash-throated flycatcher (*Myiarchus cinerascens*)	Occupancy was not influenced by noise from gas well compressors but bird vocalizations were; and individuals in areas with more noise vocalized at frequencies ~200 kHz higher. Noise levels averaged 37.4 and 56.1 dB at control and treatment sites, respectively.	Francis et al. 2011
	Song sparrows (*Melospiza melodia*)	Males shifted more energy into the higher frequencies of their vocalizations when there was more noise (total ambient background noise ranged from 54.8 – 71.3 dB).	Wood and Yezerinac 2006
	House sparrows (*Passer domesticus*)	Nests in area with large generator noise (68 dB) produced fewer young of lower body mass, and fewer recruits; females also provided young with food less often in noisy area.	Schroeder et al. 2012
	Tree swallows (*Tachycineta bicolor*)	Nestlings exposed to white noise playbacks (65 dB) had begging calls with higher minimum frequencies and narrower frequency ranges. These effects persisted in the absence of noise, suggesting that noise may influence call development. Further, when exposed to playbacks, nestlings were less likely to beg when parents arrived with food.	Leonard and Horn 2008
	Black-capped chickadee (*Poecile atricapillus*)	Noise reduced the number of individuals that could be heard, thus limiting mate choice and rival assessment.	Hansen et al. 2015
	Northern spotted owl (*Strix occidentalis occidentalis*)	Males had highest glucocorticoid response to experimentally applied motorcycle noise in May, when they are generally responsible for feeding themselves, their mates, and their nestlings.	Hayward et al. 2011
	Quail (*Coturnix coturnix*)	When quail were exposed to 116 dB of noise for 4 hours, they experienced hearing loss of up to 50 dB immediately following exposure.	Niemiec et al. 1994
	Greater sage-grouse (*Centrocercus urophasianus*)	Fecal corticosterone metabolite levels were 16.7% higher, on average, at leks where 67.6 dB of road noise was broadcast vs. control leks with no noise. Further, peak male attendance and abundance at noise-treated leks decreased by over 29% when compared to paired controls.	Blickley et al. 2012a, b

Table 2. continued.

Taxa	Species	Response	Source
Herpetofauna	Bischoff's tree frog (*Boana bischoffi*)	Advertisement call rates decreased during playbacks of road noise (played at two intensities- 65 and 75 dB) and dominant call frequency decreased when exposed to noise.	Caorsi et al. 2017
	Green frog (*Rana clamitans*), leopard frog (*R. pipiens*), gray treefrog (*Hyla versicolor*)	Call rates were significantly lower at low-noise sites (mean = 43.8 dB) than high-noise sites (mean = 73.2 dB). Further, when traffic noise was broadcast at low-noise sites, green and leopard frog vocalizations changed to having higher frequencies.	Cunnington and Fahrig 2010
	European tree frog (*H. arborea*)	Exposure to traffic playback noise (76 dB) led to increased stress hormone levels and in turn, an immunosuppressive effect.	Troianowski et al. 2017
	Wood frogs (*Lithobates sylvaticus*)	Traffic playback noise (87 dB) increased levels of glucocorticoid hormones in females. It also negatively influenced female travel towards male breeding choruses, highlighting the sublethal impacts of acoustic habitat loss.	Tennessen et al. 2014
	Grey treefrog (*Hyla chrysoscelis*)	Traffic playback noise (70 dB) resulted in female frogs taking longer to localize male calls; females were also less successful in correctly orienting to male signals.	Bee and Swanson 2007
Invertebrates	Grasshoppers (*Chorthippus biguttulus*)	Compared to males from quiet habitats, males in roadside habitats produced acoustic courtship songs with higher local frequency maximum (6-9 kHz).	Lampe et al. 2012
	Cicada (*Cryptotympana takasagona*)	Cicadas shifted the energy distribution of calling songs to higher frequencies when higher anthropogenic noise.	Shieh et al. 2012

The effects that anthropogenic noises can have on species' habitat selection, activity patterns, phenology, and physiology can culminate in decreased reproductive success. This decrease may be a consequence of limited mate choice, a reduction in pairing success, decreased provisioning rates to offspring, or a decline in offspring survival (Table 2; Francis and Barber 2013). If noise impedes the transmission of bird songs, for example, it may negatively impact mate attraction (Klump 1996; Hansen et al. 2005). If noise impedes parent-offspring communication, alternatively, it may result in young receiving food less often (e.g., if nestlings fail to beg when their parents arrive; Leonard and Horn 2012; Schroeder et al. 2012). Numerous species of birds, including eastern bluebirds (*Sialia sialis*), great tits (*Parus major*), and house sparrows (*Passer domesticus*), are known to produce fewer eggs in noisier areas (Halfwerk et al. 2011b; Kight et al. 2012; Schroeder et al. 2012). Lastly, anthropogenic noise may make it harder for females to detect and locate males, as has been documented in frogs (Bee and Swanson 2007; Tennessen et al. 2014).

FUTURE DIRECTIONS

California's Department of Food and Agriculture (CDFA) identified several potential impacts of the noises associated with cannabis cultivation in their Program Environmental Impact Report (PEIR; CDFA 2017). This noise may result from the use of irrigation pumps, diesel generators, landscaping equipment, equipment and water trucks, worker vehicles, and if a greenhouse has climate control, the heating, ventilation, and air conditioning systems.

As outlined in the PEIR, increased noise and human presence may cause substantial adverse effects on special-status terrestrial wildlife species, and use of mechanical equipment for the cultivation of cannabis may cause excessive ground borne vibration or ground borne noise levels, as well as substantial increases in ambient noise levels in the vicinity of a proposed program activity (CDFA 2017). Upon review, however, CDFA found all noise-related impacts to be "less than significant", stating that in general, the noises resulting from cannabis cultivation would be consistent with other land uses in the area (CDFA 2017). We propose, however, that the noises resulting from cannabis cultivation may differ from those associated with other land uses in the area and warrants further consideration and research.

Determining whether the noises resulting from cannabis cultivation are consistent with other land uses in the area requires an understanding of the noises' duration, loudness (i.e., decibels), and spatial location. Short-term noises from chainsaws, mowers, and vehicles may be consistent with other human-generated noises in an area; however, long-term noises from irrigation pumps, diesel generators, and climate control systems may be new. These long-term noises may adversely affect local fauna not only because they are novel, but also because they are perpetual, meaning they act as a constant impediment to the ability of the species to hear. Loudness of a noise may also play a role in determining impacts, particularly when loudness is considered in relation to ambient noise levels. A generator running at night, for example, likely has greater impacts on surrounding wildlife in a rural area, where ambient noise levels are around 20 dB, than in an urban area, where ambient noise levels are around 40 dB (Dooling and Popper 2007; CDFA 2017).

To date, most mixed-light licenses have been issued in Humboldt and Mendocino counties in northwestern California, a region of the state that is relatively undeveloped and until recently, was predominantly covered in natural vegetation (Butsic et al. 2018). This suggests that cannabis cultivation may be concentrated in rural, forested areas where the negative impacts of anthropogenic noise are likely amplified. Empirical data assessing the distribution and impacts of noises resulting from cannabis cultivation, however, are scant. Consequently, in relation to permitted cannabis cultivation in California, we encourage:

- Studies that evaluate the sound output (loudness, frequency, and duration) of cannabis growing operations in rural vs. suburban areas and how sound outputs (a) vary on a daily and annual basis, (b) compare to ambient noise conditions, and (c) compare to the sound outputs of other agricultural practices.
- Studies that assess the effectiveness of varying types of sound attenuation or insulation devices, with the goal of providing recommendations on the best devices/approaches for minimizing sound output to cannabis cultivators.
- Studies that evaluate the level of sound output (specific to cannabis cultivation) necessary to cause take, harassment, or behavioral changes in a variety of threatened and endangered species and how this varies between rural, forested habitats and suburban habitats.
- Studies assessing the call output levels (loudness, frequency, duration) and call response rates of songbirds and raptors in areas with cannabis cultivation vs. (a) areas with no human development and (b) areas with other forms of human development.
- Improving our understanding of the noises associated with cannabis cultivation and how they vary spatially, temporally, and in relation to ambient noise conditions is a critical first step in understanding how these noises may be impacting terrestrial wildlife in California and how they could be better mitigated in the future.

Author Contributions

Conceived and designed the study: LNR, ADB
Collected the data: LNR, ADB
Performed the analysis of the data: LNR, ADB
Authored the manuscript: LNR, ADB, EC
Provided critical revision of the manuscript: ADB, EC

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Examples of plastic trash on or near the banks of streams at cannabis cultivation sites in the Emerald Triangle (Humboldt, Mendocino, and Trinity Counties), CA, 2018–2019. Photo Credit: Kalyn Bocast, CDFW (top and center); CDFW staff (bottom)

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Potential impacts of plastic from cannabis cultivation on fish and wildlife resources

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Plastic is commonly used in many applications for the cultivation of cannabis. This document provides a synthesis of available scientific literature on how plastic, particularly that used in cannabis cultivation, may detrimentally affect wildlife, fish, and associated ecosystems, including entanglement and ingestion, leaching of chemicals into the environment, and alteration of soil properties.

Key words: cannabis, chemical additives, entanglement, fish, microplastics, monofilament netting, plastic, soil properties, wildlife

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Plastic is a chemically diverse group of synthetic polymer-based materials. Over 320 million tons of plastic are produced annually worldwide in sizes ranging from microplastics (< 5 mm in diameter; Barnes et al. 2009; Wagner et al. 2014; World Economic Forum 2018) to macroplastics (>20 mm in diameter). Because plastics are virtually non-biodegradable, they are mechanically broken down (e.g., physical fragmentation from weather such as hail) and are eventually released into terrestrial and aquatic ecosystems (Horton et al. 2017; Steinmetz et al. 2016; de Souza Machado et al. 2017). Given the mass production of plastic and its durability, plastic pollution has been identified as one of the most widespread and long-lasting anthropogenic changes to our planet's surface (Barnes et al. 2009). This anthropogenic change is a growing hazard for fish, wildlife, and the habitats upon which they depend.

This review provides a synthesis of available scientific literature on how plastic use in agriculture may impact wildlife, fish, and associated ecosystems to help identify the potential impact of plastic use from cannabis agriculture. The use of plastic materials in agriculture was first introduced in 1948 in the United States to cover greenhouses with cellophane (Scarascia-Mugnozza et al. 2011). The use of plastic in agriculture is now extensive and expanding. Plastic films (e.g., greenhouses, tunnels, and mulching) are used to protect crops from the environment and to create a controlled growing environment. Plastics are used to shield plants from extreme temperatures, wind, hail, wildlife damage, and to provide shading. Plastics are also used in piping, irrigation and drainage. Some reported benefits of using plastic in agricultural applications include increased yields, earlier harvests, reduction of herbicide and pesticide consumption, frost protection and water conservation, and preservation, transportation, and commercialization of food products (Scarascia-Mugnozza et al. 2011).

There is limited published information on outdoor cannabis cultivation practices. This review assumes that largely, cannabis cultivation is similar to other agricultural practices. At outdoor cannabis cultivation sites, cultivators may use, for example, plastic mulching to protect seedlings and shoots, polyvinyl chloride (PVC) pipes to transport water, plastic monofilament for plant support or erosion control, plastic netting to exclude birds and other wildlife, and an array of additional plastic products (e.g., fertilizer bags and pots). Polyolefins (i.e., plastics used for hoop houses) encompass both polyethylene (PE) and polypropylene (PP), with low-density PE being the largest component of plastic produced globally and one of the most common polymers recovered as aquatic debris (Rochman et al. 2013). Polyolefins degrade extremely slowly, meaning they can survive in the environment for 10s to 100s of years (World Economic Forum 2018).

Agricultural areas in particular, have been identified as a major entry point for plastics into continental systems (Horton et al. 2017). However, research on the impacts of plastics on the environment has predominantly focused on marine aquatic systems, with freshwater and terrestrial ecosystems only being considered in recent years (Wagner et al. 2014; Horton et al. 2017; de Souza Machado et al. 2017). Freshwater bodies often have comparable amounts of plastic to marine waters and approximately 80% of plastic pollution in the ocean comes from land via wind, direct runoff following rainstorms, and wastewater (Dris et al. 2015; Jambeck et al. 2015; Holland et al. 2016).

This review categorizes the harmful impacts of plastic use on the into three pathways: entanglement and ingestion by wildlife, leaching of harmful chemical additives into the environment, and alteration of soil properties. The review aims to serve as a starting point in documenting complex interactions between an emerging agricultural product and the environment. We have included examples from species that reside in and outside of California given many non-resident species share similar life history traits to resident species.

ENTANGLEMENT AND INGESTION BY WILDLIFE

UV radiation and temperature fluctuations fragment plastics on land while waves, wind, and UV fragment them in the ocean and freshwater lakes, creating smaller and smaller plastic particles. As the size of the plastic decreases, the number of wildlife species that could potentially ingest the plastic increases (Barnes et al. 2009; Horton et al. 2017). When plastics are ingested, they may clog feeding appendages or the digestive system, be retained in the gut, cross the gut wall into other body tissues, or be excreted at higher concentrations than when they were ingested (Barnes et al. 2009; Lwanga et al. 2017). Further, large plastic material (e.g., plastic mulch) can fragment into microplastics that are ingestible by a wider range of species, in turn facilitating their accumulation in the environment and in the food web (Barnes et al. 2009; Oehlmann et al. 2009; Steinmetz et al. 2016; Lwanga et al. 2017). In a farming landscape, for example, microplastic concentrations increased from soil to earthworm casts to chicken feces (Lwanga et al. 2017).

Wildlife species ranging from zooplankton to American robins (*Turdus migratorius*) to bull snakes (*Pituophis catenifer*) may ingest or become entangled in plastic, which may pose a considerable threat to the species (Barnes et al. 2009; Rehse et al. 2016; Gil-Delgado et al. 2017; Holland et al. 2016). Plastic that gets entangled around wildlife species' legs and feet may in time, tighten in response to swelling and can lead to necrosis of the limb (Burton and Doblar 2004). Entanglement may also result in severe lacerations, reduced mobility, or death (e.g., from strangulation or being trapped in the sun; Burton and Doblar 2004; Kapfer and Paloski 2011; Stuart et al. 2001). Table 1 includes examples from the available scientific literature of wildlife using plastic, becoming entangled in plastic, or ingesting plastic, and the effect of doing so.

Table 1. Examples of wildlife using plastic, becoming entangled in plastic, or ingesting plastic.

Taxa	Species	Effect	Source
Birds	Mourning dove (*Zenaida macroura*)	Documented becoming entangled in monofilament and then dying.	Parker and Blomme 2007
	Northern gannets (*Sula bassana*)	In two colonies of gannets, 97% of nests sampled had plastic incorporated in them including rope/line/netting (78%), plastic package strapping (12%), bags or sheets (7%) and hard plastic (3%).	Montevecchi 1991
	European coot (*Fulica atra*), mallard (*Anas platyrhynchos*), and shelduck (*Tadorna tadorna*)	There was a high prevalence of plastics in the birds' feces.	Gil-Degado et al. 2016
	Osprey (*Pandion haliaetus*)	Nestlings can become entangled in the bailing twine that has been incorporated into their nests and perish.	Blem et al. 2002
	Mallard (*A. platyrhynchos*), American black duck (*A. rubripes*), and common eider (*Somateria mollissima*)	Plastic was found in the stomachs of 46% of mallards, 7% of black ducks, and 2% of eiders analyzed.	English et al. 2015
	American crow (*Corvus brachyrhynchos*)	85% of crow nests along an urban to agricultural gradient contained anthropogenic material; the amount of material was higher in nests in agricultural areas than urban areas; all entangled nestlings failed to fledge.	Townsend and Barker 2014
	Ducks, geese, American robins (*Turdus migratorius*), and Eastern bluebirds (*Sialia sialis*)	Monofilament can become entangled around the distal legs and feet, where it tightens in response to swelling. This can lead to necrosis of the limb and eventual amputation.	Blem and Doblar 2004

Taxa	Species	Effect	Source
	California condor (*Gymnogyps californianus*)	Ingestion of anthropogenic garbage, including plastic, has slowed the development of feathers in some nestlings and resulted in the death of others; nestlings may be physiologically less able to regurgitate pellets or other indigestible material than adults.	Mee et al. 2007
	Great tit (*Parus major*)	Appeared to preferentially seek out anthropogenic material for nests; more anthropogenic material was associated with lower general arthropod diversity and higher levels of Siphonaptera (fleas).	Hanmer et al. 2017
Herpetofauna	Leatherback sea turtle (*Dermochelys coriacea*)	Ingested plastic can result in esophageal and gastrointestinal blockage and death.	Starbird and Audel 2000
	Coachwhips (*Masticophis flagellum*) and bullsnake (*Pituophis catenifer*)	Have become entangled in plastic netting, sometimes leading to death (e.g., from overheating after being entrapped in full sunlight).	Stuart et al. 2001
	Common gartersnake (*Thamnophis sirtalis*), northern watersnake (*Nerodia sipedon*), Western fox snake (*Pantherophis vulpinus*)	Have been found entangled in plastic netting.	Kapfer and Paloski 2011
Invertebrates	Earthworms (*Lumbricus terrestris*)	In a lab, there was a significant reduction in growth rates when exposed to microplastics; mortality was also observed with mortality increasing as concentration of microplastics increased; there were negative effects on burrow construction.	Lwanga et al. 2016
	Earthworms	Earthworm casts contained concentrated amounts of microplastics. This is a consequence of their direct ingestion of the soil.	Lwanga et al. 2017
	Earthworm (*Eisenia andrei*)	In a lab, worms were exposed to different concentrations of microplastics. There were no significant effects on survival or reproduction but there was damage to the gut and immune system.	Rodriguez-Seijo et al. 2017
	Zooplankton (*Daphnia magna*)	Ingestion of plastic particles at high doses lead to immobilization.	Rehse et al. 2016
Fish	Freshwater and marine teleost fishes	In natural settings, microplastics have been found to be ingested by several fish species, no matter the size, life stage or life history.	Hoss and Settle 1989; Eerkes-Medrano et al. 2015; Collicutt et al. 2019

LEACHING OF HARMFUL CHEMICAL ADDITIVES

Chemical additives such as Bisphenol-A (BPA), polybrominated diphenyl ethers (PBDE), or phthalate acid esters (PAE) are added to plastics to increase their functionality (e.g., elasticity, rigidity, and UV stability). Over half of all plastics are associated with hazardous monomers, additives, and/or chemical byproducts (Rochman et al. 2013). These hazardous monomers and additives are weakly bound or not bound at all to the polymer molecule (i.e., to the plastic) meaning that over time, they will leach out of the plastic and into surface waters, wastewater, groundwater, sediment, and soil (Clara et al. 2010; Steinmetz et al. 2016; Horton et al. 2017). Leached chemical additives may be carcinogenic or toxic and many function as endocrine disruptors that negatively impact developmental,

Table 2. Examples of how the leaching of chemical additives from plastics may impact wildlife.

Taxa	Species	Effect	Source
Mammals	Rats and mice	In a lab, adult exposure to BPA affected the male reproductive tract; developmental exposure affected the brain and metabolic processes.	Richter et al. 2007
	Rats	In a lab, high doses of DEHP led to rapid and severe changes in the testes of adult male rats and adverse responses in females (following pre- and post-natal exposure).	Talsness et al. 2009
	Mice, guinea pigs, and ferrets	In a lab, exposure to phthalates sometimes induced testicular injury.	Oehlmann et al. 2009
Herpto-fauna	African clawed frog (*Xenopus laevix*)	In a lab, BPA exposure led to teratogenic effects like crooked vertebrae, abnormal development of head and abdomen, and death of cells in the central nervous system.	Oka et al. 2003
	Moor frog (*Rana arvalis*)	In a lab, exposure to DEHP via sediment resulted in decreased successful hatchings with increasing concentrations.	Larsson and Thurén 1987
	Japanese wringled frog (*Rana rugosa*)	In a lab, DBP exposure caused delayed gonadal development in male tadpoles.	Ohtani et al. 2000
Inverte-brates	Ramshorn snails (*Marisa cornuarietis*)	In a lab, exposure to BPA caused superfeminization syndrome (i.e., additional sex organs, enlarged accessory sex glands, enhanced egg production) outside of spawning season and increased female mortality.	Oehlmann et al. 2000

Table 2. continued.

Taxa	Species	Effect	Source
	Crickets	In a lab, ingestion of polyurethane foam led to bioaccumulation of chemical additives in crickets.	Gaylor et al. 2012
	Lugworms (*Arenicola marina*)	In a lab, worms that were fed polystyrene with sorbed chemical additives bioaccumulated the chemical additives.	Besseling et al. 2013
	Annelid (*Capitella capitata*)	In a lab, exposure to BPA caused premature metamorphosis of larvae.	Biggers and Laufer 2004
	Chironomid larvae (*insect*)	In a lab, exposure to BPA caused delayed larval emergence.	Watts et al. 2003
Fish	Carp, fathead minnow, rainbow trout	In a lab, BPA exposure had feminizing effects in vivo, induced synthesis of zona radiata proteins, and induced alterations in gonadal development and gamete quality.	Oehlmann et al. 2009
	Fathead minnow (*Pimephales promelas*)	In a lab, exposure to BPA increased percentage of spermatocytes.	Sohoni et al. 2001
	Common carp (*Cyprinus carpio*)	In a lab, exposure to BPA caused alterations in the gonadal structure of males and in some instances, intersex.	Oehlmann et al. 2009
	Common carp	In a lab, exposure to DEP caused changes in general behavior.	Barse et al. 2007
	Common carp	in lab, exposure to BPA caused gonad structural changes in males and decreased estrogen to androgen ratios in blood.	Mandich et al. 2007
	Brown trout (*Salmo trutta f. fario*)	In a lab, exposure to BPA caused reduced sperm quality, delayed ovulation in females, and inhibition of ovulation in females.	Lahnsteiner et al. 2005
	Brown trout (*Salmo trutta f. fario*)	In a lab, low exposure to BPA caused reduced sperm quality and delayed ovulation; higher exposure caused complete inhibition of ovulation.	Lahnsteiner et al. 2005
	Atlantic salmon (*Salmo salar*)	In a lab, exposure to DEHP in food during early life resulted in a small incidence of intersex.	Norman et al. 2007
	Three-spined stickleback (*Gasterosteus aculeatus*)	In a lab, exposure to phthalates caused alterations in shoaling and feeding behavior.	Wibe et al. 2004
	Fish in general	Phthalates have been detected in wild fish and have been found to bioconcentrate in the body tissues of some fish.	Oehlmann et al. 2009

metabolic, and reproductive processes (Richter et al. 2007; Oehlmann et al. 2009; Talsness et al. 2009; Flint et al. 2012; Lü 2018; Teuten et al. 2009). The adverse impacts of chemical additives can be even more acute in developing organisms given their greater sensitivity to drug and chemical exposure (Talsness et al. 2009). Exposure to very low doses of BPA (i.e., doses lower than those studied for toxicological risk assessment purposes) has been found to negatively impact experimental mammals, crustaceans, aquatic insects, and fish (Richter et al. 2007; Oehlmann et al. 2009). Phthalates like diethyl phthalate (DEP), diethylhexyl phthalate (DEHP), and dibutyl phthalate (DBP), alternatively, are known to negatively affect reproduction, to impair development, and/or to induce genetic aberrations in wildlife groups like molluscs, crustaceans, and amphibians (Oehlmann et al. 2009). Smaller-sized plastic has a greater likelihood of leaching chemical additives into the environment, owing to their larger surface to volume ratio (de Souza Machado et al. 2017). Table 2 includes examples from the available scientific literature of how the leaching of chemical additives from plastics may impact wildlife.

ALTERATION OF SOIL BIOGEOCHEMISTRY AND BIOPHYSICAL PROPERTIES

Plastic placed on top of soil (e.g., plastic mulch or monofilament erosion control), as well as other plastic used in cannabis cultivation (e.g., fertilizer bags and pots) have the potential to alter the soil's biogeochemistry and biophysical properties (Steinmetz et al. 2016; Horton et al. 2017; de Souza et al. 2018). Plastic mulches, for example, may induce changes in the soil microbial community. They may modify microclimate conditions (e.g., temperature and moisture), which in turn may increase biological degradation of litter and soil organic matter that in turn, deplete soil nutrients like carbon, alter root or soil fungi relationships, and decrease the abundance of ecosystem engineers like earthworms and nematodes (Steinmetz et al. 2016). Plastic mulches may also impact cannabis cultivation sites by enhancing water runoff into furrows or un-mulched areas. This has multiple impacts including increasing the areas' susceptibility to soil erosion, decreasing soil structural stability, and elevating pesticide loads in these bare ground areas (Steinmetz et al. 2016).

Plastic mulches, plastic monofilament, and the array of other plastic products used on cannabis cultivation sites will fragment over time (e.g., by UV radiation and temperature fluctuations) if they are not cleaned up on a regular basis. Soils will then function as the long-term sink for plastic fragments and debris left behind, with plastics persisting upwards of 100 years in the soil due to low light and oxygen contents (Horton et al. 2017; de Souza et al. 2018). Plastic particles can alter the soil's biophysical environment by changing: 1) soil bulk density- plastics are often less dense than many natural minerals predominant in soils, 2) soil moisture and evapotranspiration – some types of plastic can increase soil's water holding capacity while others can decrease it, 3) microbial activity, and 4) invertebrate activity (Lwanga et al. 2017; de Souza et al. 2018; Zhu et al. 2018). Exposing earthworms (*Lumbricus terrestris*) to microplastics, for example, changed the worms' burrow structure and in turn, soil aggregation and function (Lwanga et al. 2017). Microplastics in soil also impacted the activity of springtails (a hexapod), which then effected the springtails' gut microbiomes and ultimately their growth and reproduction (Zhu et al. 2018).

CONCLUSION

The use of plastic in agriculture is not unique to cannabis cultivation, but information on cannabis cultivation practices in California to date is limited. Although there has not been a formal survey of the use of plastic in cannabis cultivation, it is commonly believed to be widespread. In an online survey conducted in 2018, cannabis growers indicated that most cannabis (41%) was produced outdoors (open air, sunlight), followed by greenhouse (25%; partial or full sunlight) (Wilson et al. 2019). Both methods likely use plastic piping for irrigation and plastic monofilament netting as scaffolding to support plants. Many cannabis growers use hoop houses- greenhouses constructed by placing polyethylene plastic over a PVC arch frame. There are many important gaps in information regarding cannabis cultivation practices that, when answered, will help our understanding of how the environment may be affected by the use of plastic. This review assumes that cannabis cultivation practices are comparable to other agricultural practices. However, further research is required to determine if this assumption is valid. More information is needed on the type, amount, duration, and timing of plastic use on cannabis farms. Research on disposal methods of used plastic is essential. Agricultural plastic products are difficult to collect, recycle, and reuse (Steinmetz et al. 2016). As more information is gathered on the use of plastics in cannabis cultivation, it will be important to continue to synthesize the effects of such materials on wildlife, fish, and associated habitat. This will allow for the development of science-based best management practices to mitigate or avoid detrimental effects.

Author Contributions

Conceived and designed the study: LNR, MM, EF, ADB
Collected the data: LNR, EF, ADB
Performed the analysis of the data: LNR, EF, ADB
Authored the manuscript: LNR, MM, EF, ADB, EC
Provided critical revision of the manuscript: MM, EF, ADB, EC

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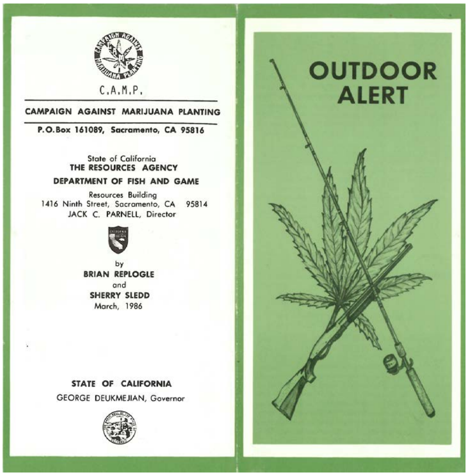
California Fish and Wildlife, Cannabis Special Issue; 133-134; 2020

CDFW's history with cannabis cultivation

In 1986, when the brochure, below, was created, the multi-agency entity Campaign Against Marijuana Planting (CAMP; created in 1983) , that the California Department of Fish and Game (CDFG) was a part of, was actively involved in raiding large-scale cannabis grows, especially in the North Coast. At that time, all grows were illegal and most grows were outdoor operations. In response to intensive CAMP operations, many growers became reluctant to grow on their properties because enforcement agencies would regularly confiscate land, homes, vehicles, and other personal property from illegal growers. This reality compelled many growers to establish rogue grows on public land as a means of reducing the risk of losing personal assets. As a result, hunters and anglers had greater risk of coming across illicit grows while on public lands; hence, the development of the brochure. Despite this potential risk, CDFG's Law Enforcement received only a few calls from hunters and anglers regarding trespass grows on public lands. Workers hired to maintain the trespass grows rarely confronted hunters and anglers. Cartel grows and the resulting higher risk of confrontation was not common at that time.



C.A.M.P.

CAMPAIGN AGAINST MARIJUANA PLANTING

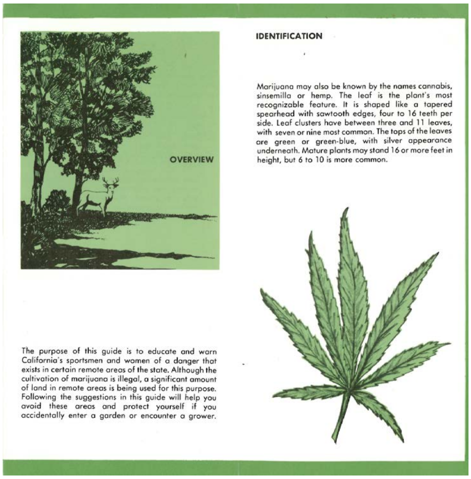
P.O.Box 161089, Sacramento, CA 95816

State of California
THE RESOURCES AGENCY
DEPARTMENT OF FISH AND GAME
Resources Building
1416 Ninth Street, Sacramento, CA 95814
JACK C. PARNELL, Director

by
BRIAN REPLOGLE
and
SHERRY SLEDD
March, 1986

STATE OF CALIFORNIA
GEORGE DEUKMEJIAN, Governor

OUTDOOR ALERT



OVERVIEW

The purpose of this guide is to educate and warn California's sportsmen and women of a danger that exists in certain remote areas of the state. Although the cultivation of marijuana is illegal, a significant amount of land in remote areas is being used for this purpose. Following the suggestions in this guide will help you avoid these areas and protect yourself if you accidentally enter a garden or encounter a grower.

IDENTIFICATION

Marijuana may also be known by the names cannabis, sinsemilla or hemp. The leaf is the plant's most recognizable feature. It is shaped like a tapered spearhead with sawtooth edges, four to 16 teeth per side. Leaf clusters have between three and 11 leaves, with seven or nine most common. The tops of the leaves are green or green-blue, with silver appearance underneath. Mature plants may stand 16 or more feet in height, but 6 to 10 is more common.

OUTDOOR ALERT: Produced in March, 1986 by Campaign Against Marijuana Planting (CAMP).

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Some counties present a higher risk than others because the remoteness that attracts hunters and anglers also aids the grower, who is trying to avoid detection. We recommend you contact the sheriff's department or Department of Fish and Game before you hunt or fish in c;m unfamiliar area. These agencies can identify places that should be avoided.

This map indicates the 10 counties with the highest incidence of marijuana cultivation based on a 1985 report of Campaign Against Marijuana Planting (C.A.M.P.). Take precautions when planning outdoor trips in these regions.

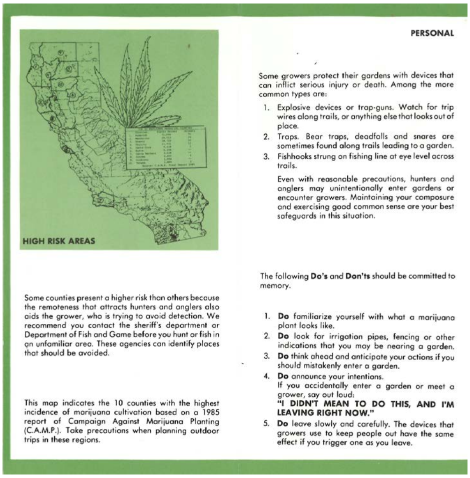
Some growers protect their gardens with devices that can inflict serious injury or death. Among the more common types are:

1. Explosive devices or trap-guns. Watch for trip wires along trails, or anything else that looks out of place.
2. Traps. Bear traps, deadfalls and snares are sometimes found along trails leading to a garden.
3. Fishhooks strung on fishing line at eye level across trails.

Even with reasonable precautions, hunters and anglers may unintentionally enter gardens or encounter growers. Maintaining your composure and exercising good common sense are your best safeguards in this situation.

The following Do's and Don'ts should be committed to memory.

1. Do familiarize yourself with what a marijuana plant looks like.
2. Do look for irrigation pipes, fencing or other indications that you may be nearing a garden.
3. Do think ahead and anticipate your actions if you should mistakenly enter a garden.
4. Do announce your intentions. If you accidentally enter a garden or meet a grower, say out loud: "I DIDN'T MEAN TO DO THIS, AND I'M LEAVING RIGHT NOW."
5. Do leave slowly and carefully. The devices that growers use to keep people out have the same effect if you trigger one as you leave.
6. Don't panic. Remember that common sense and composure are your best protection.
7. Don't be a hero. Challenging a grower or attempting to destroy a garden is extremely dangerous. Nearly all growers are well armed and have the advantage of knowing the area. Even if you don't see anyone, there is a chance your presence is known.
8. Don't approach a garden intentionally. Curiosity can be dangerous. If you see a garden or any indications of one, stay away.



HIGH RISK AREAS

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PERSONAL

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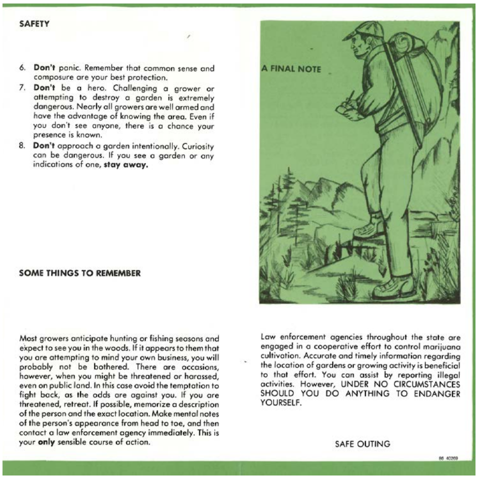
The following **Do's** and **Don'ts** should be committed to memory.

1. **Do** familiarize yourself with what a marijuana plant looks like.
2. **Do** look for irrigation pipes, fencing or other indications that you may be nearing a garden.
3. **Do** think ahead and anticipate your actions if you should mistakenly enter a garden.
4. **Do** announce your intentions. If you accidentally enter a garden or meet a grower, say out loud: **"I DIDN'T MEAN TO DO THIS, AND I'M LEAVING RIGHT NOW."**
5. **Do** leave slowly and carefully. The devices that growers use to keep people out have the same effect if you trigger one as you leave.

SOME THINGS TO REMEMBER

Most growers anticipate hunting or fishing seasons and expect to see you in the woods. If it appears to them that you are attempting to mind your own business, you will probably not be bothered. There are occasions, however, when you might be threatened or harassed, even on public land. In this case avoid the temptation to fight back, as the odds are against you. If you are threatened, retreat. If possible, memorize a description of the person and the exact location. Make mental notes of the person's appearance from head to toe, and then contact a law enforcement agency immediately. This is your only sensible course of action.

Law enforcement agencies throughout the state are engaged in a cooperative effort to control marijuana cultivation. Accurate and timely information regarding the location of gardens or growing activity is beneficial to that effort. You can assist by reporting illegal activities. However, UNDER NO CIRCUMSTANCES SHOULD YOU DO ANYTHING TO ENDANGER YOURSELF.



SAFETY

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A FINAL NOTE

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SAFE OUTING

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Authors: Brian Replogle and Sherry Sledd

INFORMATION FOR AUTHORS

The California Fish and Wildlife Journal (CFWJ) is a peer-reviewed, scientific journal focused on the biology, ecology, and conservation of the flora and fauna of California and surrounding areas, and the northeastern Pacific Ocean.

Submissions guidelines (PDF) for the Journal have been updated (July 2019).

The California Fish and Wildlife Journal accepts manuscripts in the following categories:

- Original research papers
- Research notes
- Review papers
- Book reviews
- Commentaries and Essays

Manuscripts must be submitted by e-mail following directions provided in the link: https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=171113&inline. The journal standard for style is consistent with the Council of Science Editors (CSE) Style Manual. Instructions in the CFWJ guidelines supersede the CSE Style Manual where differences exist between formats. Please follow these formatting guidelines carefully. Manuscripts that do not conform to the guidelines will be returned for revision.

Authors of manuscripts that are accepted for publication will be invoiced for charges at the rate of $50 per printed page shortly after page proofs are distributed.* Authors should state acceptance of printing charges in their cover letters. The corresponding author will receive a PDF file of the publication without additional fees and may distribute copies without restriction.

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Front. Cannabis cultivation in Humboldt County on a plot of cleared forestland. Photo Credit: Scott Bauer, CDFW

Back. Bud of Cannabis strain “Afgan Kush” cultivated in Lake County, California. Photo Credit: Margaret Mantor, CDFW.

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