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Urban forest cover and ecosystem service response to fire varies across California communities

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ABSTRACT

Urban fires that result from wildfires are an emerging, extreme event affecting communities and urban forests globally. However, much of the fire effects on urban ecosystems literature is primarily focused on Wildland-Urban Interface (WUI) areas, correlates of building loss, risk mitigation, and wildland vegetation and fuels. Three recent urban fires in California USA provided an opportunity to explore these effects on urban forests: the 2017 Tubbs (Santa Rosa) and Thomas (Ventura) fires and the 2018 Camp fire that burned Paradise. Accordingly, we analyzed pre- and post-fire neighborhood level urban tree cover (nUTC) change over 5 years using Sentinel and LiDAR data (10 m resolution). Then, we explored the effects of fire severity on changes in several regulating ecosystem services (e.g., carbon, air pollution, stormwater). Findings from the Rapid Assessment of Vegetation Condition after Wildfire severity processes and other geospatial datasets show that fire effects were patchy with fire severity within neighborhoods ranging from unburned to extreme. We found that ~5 years after the fires, and relative to adjacent non-fire affected neighborhoods, Ventura and Santa Rosa's nUTC is recovering to pre-fire levels while Paradise's nUTC is being consistently lost over time. Ventura and Santa Rosa had 20–25 % of their fire affected area outside established WUI boundaries. However, local-scale urban tree cover and ecosystem service supply are lagged, and recovery time depends on the surrounding biome and socio-ecological context. In inland, higher elevation communities – like Paradise – the recovery of nUTC and the associated ecosystem services might materialize over a much longer timeframe. Our study provides a roadmap to assess the response of urban wildfire-affected tree cover and ecosystem services across space and time. It is also one of the first assessments of fire effects on urban ecosystems and communities outside of WUI boundaries that are now experiencing increased threat from wildfires globally.

1. Introduction

Urban fires caused by wildfires are an emerging and novel extreme event affecting communities and cities in many regions of the world (UNEP, 2022; Calkin et al., 2023; Wasserman and Mueller, 2023). The environmental, socioeconomic, and ecological effects of wildfire are not only devastating natural ecosystems, but now urban fires are a disaster affecting human lives and wellbeing in communities, cities, and towns. In recent years, highly urbanized human settlements are increasingly being exposed to, and affected by, catastrophic fires (Yadav et al., 2023; Troy et al., 2022). Places such as Australia, southern Europe, and western USA, have historically experienced wildfires in and near highly populated, densely built urban communities (Jenerette et al., 2022; Wang et al., 2022). However, more recently places such as Chile, northern Europe, United northern areas of Canada and the eastern US are experiencing such events (Bento-Gonçalves and Vieira, 2020).

As of 2023, a fire in Hawai'i USA devastated a community and resulted in over 100 civilian fatalities and thousands had to evacuate their communities in several urban areas in western Canada and the island of Rhodes, Greece (Calkin et al., 2023; Lothian-McLean, 2023). As these communities recover and rebuild following urban fires, timely information is needed on the post-fire impacts to urban forests and their

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socio-ecological dynamics across time and space. Indeed, local and regional decision-makers often lack information regarding fire caused changes to urban forest cover, structure, and composition dynamics and how these changes might affect ecosystem services. In this contribution, we provide new information to narrow this information gap by assessing the spatial and temporal effects of recent fires on urban forest cover, ecosystem services and post-fire trajectories in three urban communities in California USA.

Measures of remotely sensed forest cover in both Wildland-Urban Interface (WUI) and urban contexts can be a first step towards understanding the emerging spatio-temporal effects of fire on urban forests. The changes in wildland and WUI forest cover caused by wildfire have been measured using satellite-based imagery (Lentile et al., 2006; Hudak et al., 2007; Masek et al., 2013). For example, Schug et al. (2023) mapped WUI extent at a global scale using building area and wildland vegetation data. The advent of new remote sensing systems, data, classification and cloud-based analytical techniques, and emerging processes platforms has further enabled potential opportunities in regional and large-scale forest change monitoring (Kurbanov et al., 2022; Schug et al., 2023). Remote sensing methods and data including high resolution airborne imaging spectroscopy, LiDAR and multi-spectral satellite images are now widely used to study the recovery rate of forests after wildfire (Meng et al., 2018).

However remote sensing-based information focusing on fire-affected forest cover and burn severity in urban land use-covers, to our knowledge is rare. Conversely, there are available studies in the urban ecology and urban forestry literature that have studied urban tree cover (UTC) dynamics in urban areas located in different biomes and climates (De la Barrera and Henríquez, 2017; Healy et al., 2022; Nowak and Greenfield, 2020; Roman et al., 2021). Indeed, in urban forest management and planning, UTC is a common indicator for assessing the structure, composition, distribution, and dynamics of urban forests (Ossola and Hopton, 2018), their ecosystem services (Dobbs et al., 2011; McDonald et al., 2020) as well as the socio-cultural drivers affecting their structure and change over time (Locke et al., 2022; Ossola et al., 2021).

The above literature shows that considerable research has been done to study the impacts of wildfire in WUI and forested areas and their management pre- and post-fire (Lentile et al., 2007; Nauslar et al., 2018; van Wagtendonk et al., 2018; Wang et al., 2022). Studies have also documented the relationship between wildfire damage to buildings (Knapp et al., 2021; Syphard et al., 2007 and 2019; Troy et al., 2022), plant flammability characteristics in WUI contexts (White and Zipperer, 2010), and vegetation and fuel treatment effects on various ecosystems (Bento-Gonçalves and Vieira, 2020; Jenerette et al., 2022). Yet, little is known about how to address these same issues in urban landscape where silvicultural, ecological, and arboricultural factors interact in complex ways with biophysical, socioeconomic and technical practices unique to urban areas (Roman et al., 2014; Nitschke et al., 2017).

Trees in urban areas, for example, are typically managed for their ecosystem services (Dobbs et al., 2011) and on an individual basis using arboricultural techniques to address issues that are often unique to the urban environment (e.g., pruning for tree structure, clearance, and aesthetics, risk reduction; Ball, 2011). Thus, when wildfires spread from WUI areas to the urban landscape, complex issues arise pertaining to – not only public health and safety and damage to property – but also the management of trees (i.e., retain, remove, or restore fire damaged trees, residual risk; Gilbert and Brack, 2007). But, there is little information on urban fire effects on urban trees and shrubs, specifically their survivability, assessing fire damage, and species-specific response to severity and how this affects the supply of ecosystem services and disservices (Escobedo et al., 2011).

Therefore, the aim of this study is to assess the spatial and temporal effects of urban fire on tree cover, ecosystem services, and post-fire trajectories in three urban communities in California USA affected by fires during 2017–2018. The specific objectives are to first, analyze fire effects on urban tree cover within the footprints of the 2017 Tubbs (City of Santa Rosa) and Thomas (City of Ventura) fires and the 2018 Camp fire (Town of Paradise) in California. Second, we explore potential changes in ecosystem services to these communities due to urban forest cover changes and how these changes can support future urban forest management decisions. And third, we discuss the distinction in management between urban fires versus Wildland-urban interface wildfires as related to urban forests. Findings will then be used to discuss future research implications for the disciplines of urban forestry, arboriculture, urban ecology, fire ecology, and to contribute to the *urban fire* versus *wildfire* discourse (Calkin et al., 2023).

2. Methods

Given the recent and unprecedented impacts of urban fires to urban ecosystems, we propose an approach to assess post-fire changes to UTC and ecosystem services using available data from these case studies. The approach and results can then be used to develop more advanced and rapid methods for mapping and assessing post-fire impacts to urban forests. It can also serve to better understand the socio-ecological impacts to urban forests and their trajectories following more frequently occurring and widespread urban fires (Calkin et al., 2023).

2.1. Study areas

The study areas are the cities of Ventura and Santa Rosa and Town of Paradise, all in California USA, which were affected by the Thomas, Tubbs, and Camp Fires, respectively (Fig. 1). Ventura, Santa Rosa, and Paradise are classified, according to climate and ecoregion characteristics as being in the Southern California Coast, Northern California coast, and Interior West climate zones, respectively (McPherson et al., 2017). In general, climate in coastal Ventura and inland Santa Rosa is Mediterranean and is characterized by dry, warm summers and precipitation during the milder winters (van Wagtendonk et al., 2018). Ventura's surrounding vegetation types are coastal sage (e.g., *Artemisia spp., Ceanothus spp., Baccharis spp.*) while Santa Rosa's peri-urban vegetation is primarily *Quercus spp*. woodlands and shrublands with areas of coniferous forests and agricultural pastures (Keely and Syphard, 2019). The climate in Paradise, located on the western slope of the northwestern Sierra Nevada, is also Mediterranean but differs slightly due to its higher elevation and inland location. Vegetation types in peri-urban areas and remnant forest patches inside the city, are characterized by conifer and oak forest types (e.g., *Pinus spp., Pseudotsuga menziesii; Calocedrus decurrens, Abies spp*; Knapp et al., 2021).

The 2017 population in Santa Rosa was approximately 180,000 (1600 persons/km^2) and 109,000 (1300 persons/km^2) in Ventura with little change through 2020 (US Census Bureau, 2022). Paradise's 2017 population was approximately 26,000 but the post-fire displacement has resulted in a five-fold population drop to about 5000 (94 persons/km^2) in 2020 (US Census Bureau, 2022). In general, all three urban forests were affected by wind-driven wildfires in conjunction with urban fire conflagration factors (e.g., abundant fuel quantity and connectivity) associated with the urban environment including: high housing density, flammable building construction practices, and close structural proximity to ignitable vegetation and ornamental features) as opposed to wildland fuel-driven wildfire factors (Keeley and Syphard, 2019). Moreover, each of these communities had experienced multiple years of drought resulting in very dry conditions with the abundant, connected, fuels experiencing moisture conditions receptive to burn (Nauslar et al., 2018).

Our three fire-affected urban forest landscapes were widely separated geographically across California (Fig. 1). In Northern California, the Tubbs fire started on October 8, 2017 and burned into Santa Rosa city limits in the early morning of October 9. The fire moved rapidly into the city limits driven by high velocity "diablo winds" moving roughly 12 miles in the first three hours (Nauslar et al., 2018). The fire continued to move rapidly through the city burning through the neighborhoods of

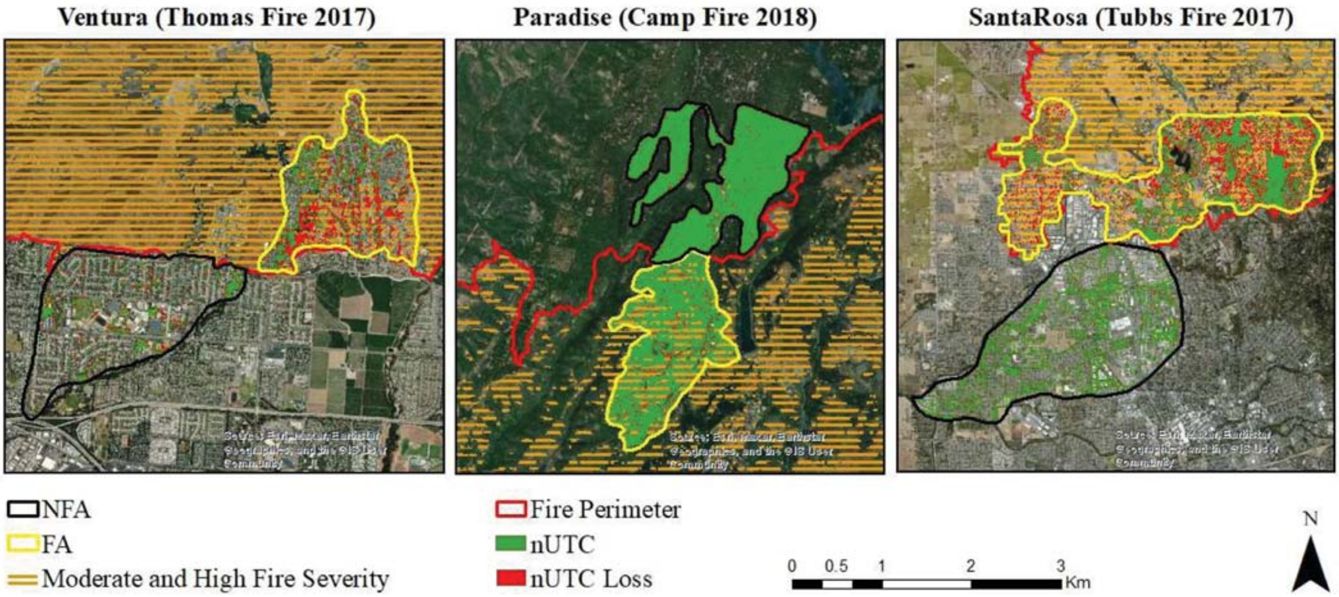

Fig. 1. Neighborhood urban tree cover (nUTC) and Rapid Assessment of Vegetation (RAVG) moderate and high fire severity classes overlaid on post-fire 1 m satellite imagery in the three pairs of fire affected (FA) and non-fire affected neighborhoods (NFA) within the Thomas, Tubbs, and Camp Fire study areas.

Mark West, Fountain Grove, and Coffee Park in addition to a local high school, several shopping centers, crossed a four-lane highway, and destroyed a large department store. Thousands of residents were evacuated, 5643 structures were destroyed and 22 people lost their lives (McNamara and Mell, 2022).

Our second landscape was affected by the Thomas fire that started on December 4, 2017 in southern California and burned into the City of Ventura destroying 500 homes the first night. The Thomas fire burned over a larger area than the Tubbs fire (114,078 vs 14,896 ha) but a smaller proportion of the area burned was in WUI and urban neighborhoods. Although 8500 firefighters were mobilized to protect Ventura and the adjacent coastal communities, the wildfire driven by strong and persistent Santa Ana winds, moved rapidly across the landscape forcing the evacuation of 104,500 residents and destroyed over 1000 structures (Nauslar et al., 2018). The Thomas fire resulted in 22 fatalities: 2 deaths were directly attributed to the fire, and 20 people lost their lives due to the post-fire debris flow in nearby Montecito California (Nauslar et al., 2018). The third landscape was affected by the 62,000 ha Camp Fire that started northwest of Paradise, California on November 8, 2018. This is the deadliest urban fire in California history and was also wind driven and moved rapidly into the Towns of Paradise, Concow, Magalia, and Butte Creek destroying more than 18,000 structures; nearly 95 % of Paradise and Concow's structures alone, and caused 85 civilian deaths (Knapp et al., 2021; Troy et al., 2022).

2.2. Remote sensing methodology

The California Forest Observatory (2020) data (CFO hereafter) is a 10 m spatial resolution forest monitoring system that uses airborne LiDAR and satellite datasets to map the drivers of wildfire behavior (e.g., vegetation metrics) in the United States. Although <10 m resolution data could be used for individual tree and parcel-level analyses, CFO's use of Sentinel and LiDAR data make it more suitable to map patterns of forest structure (i.e., tree height and canopy cover) for neighborhood and regional-level analyses. The CFO's forest structure metrics are derived from LiDAR data are only available for a few areas in California, therefore according to CFO, deep learning models were used on the satellite imagery to map these forest metrics statewide. The model performance was found to have a root mean squared error of 0.91 and 0.86 for canopy cover and tree height, respectively, and 1.97 m mean absolute error for tree height (California Forest Observatory, 2020).

Canopy height is generally defined as the distance between the ground and the top of the forest canopy, while urban forest or canopy cover (i.e., UTC) is the horizontal cover fraction occupied by the tree canopy and values range from 0–100 % (Nowak and Greenfield, 2020). Therefore, with an availability of high quality CFO forest structure spatial data in California, two sets of forest canopy-fuel metrics, canopy height and cover, were used since they are particularly appropriate for our study aim in that it can differentiate between tall shrubs and trees that are common in urban areas. Accordingly, tree and shrub cover were differentiated based on a canopy height threshold of 3 m (Blood et al., 2016); specifically, canopy heights from 3–80 m were defined as tree cover while heights <3 m were shrub cover. As such, our 3 m canopy height is a proxy for the amount of forest, non-shrub, fuels and foliage that may be consumed in a UTC-level crown fire.

A canopy height of >3 m was first extracted and used to differentiate shrubs from trees and as a mask on canopy cover was used to derive tree cover. Subsequently, the CFO tree cover data were then used to estimate detailed UTC loss in the selected pairs of neighborhoods and in high to moderate fire severity zone of the three selected fire perimeters in California during 2016–2020. Sections 2.2 and 2.3 will explain how fire severity was mapped. Further, spatial fire perimeter data from 2011–2020 compiled by Cal Fire's Fire and Resource Assessment Program (FRAP; www.frap.fire.ca.gov) was used to select fire affected nUTC loss in the three areas of interest.

We estimated UTC loss exclusively due to fire by analyzing fire-affected neighborhoods characterized by residential, commercial, industrial, and urban green space (i.e., parks, vacant areas, shrubland and forest) land use covers and where tree canopy cover ranged from 0 to 100 % and where it was greater than 65 % of a 30-meter pixel. This accounted for non-urban land covers (e.g., shrublands, WUI forests, agricultural areas, ex-urban housing) in the US Census defined urban areas affected by the Camp, Thomas, and Tubbs Fire perimeters. For example, US Census defined urban areas in Ventura affected by the 2017 Thomas fire included hills and steep topography with no urban land use-covers. Conversely, urban land use-covers in Santa Rosa and Paradise affected by the Tubbs and Camp fires, aligned well with existing city planning and zoning spatial information.

Accordingly, urban neighborhood in Paradise and Ventura were delineated by intersecting spatial data using 3 steps: 1) existing urban land cover class data from the 2016 National Land Cover Dataset, 2) available district council vector boundaries (available on www.map.cityofventura.net and resilient zone areas of Santa Rosa city; www.srcity.org), and 3) visual interpretation of high-resolution images on Google Earth. Step number 3 was necessary for Santa Rosa and Ventura, because only certain neighborhoods were affected by urban fire, therefore nUTC was adjusted and modified according to the building clusters interpreted on Google Earth. All the previously mentioned pre- and post-fire spatial layers were then used to extract nUTC, respectively, for the three neighborhoods affected by the three urban fires.

2.3. Determining areas of fire severity affected neighborhood-level urban tree cover

Urban tree cover can change over space and time due to several reasons including land use changes, urbanization, drought, policies, pests/diseases as well as fire (Healy et al., 2022; Ossola and Hopton, 2018). Similarly, forests and neighborhoods burn heterogeneously ranging from patches of unburned to high severity. Fire severity has been studied when assessing damage to wildland and WUI forests (Eidenshink et al., 2007). High fire severity or the near or total consumption of biomass and forest cover (Qi et al., 2022) as well as moderate fire severity can consume >30 % of the vegetation (Keeley, 2009). These high and moderate severities can have prolonged temporal effects on restoration needs and successional trajectories of forests and their ecosystem services (Rother et al., 2023; van Wagtendonk et al., 2018). In contrast, low-severity fires can have negligible effects on individual tree biometric attributes and stand-level function and subsequent services. For example, forest mid-story vegetation regrowth was found to be 22–40 % lower in fire affected area with low and moderate severity as compared to areas with high fire severity (Gordon et al., 2017).

Fire severity can be measured directly by on the ground plot sampling or indirectly using remotely sensed satellite earth observations such as Landsat. For example, the Rapid Assessment of Vegetation Condition after Wildfire (RAVG) is a methodology to indirectly measure vegetation change across large landscapes due to burning (e.g. fire severity) using repeated Landsat imagery (Miller and Thode, 2007). The RAVG process measures vegetation changes due to wildfire by comparing pre- and post-fire multispectral satellite data from Landsat earth observations and calculates the relativized differenced normalized burn ratio (RdNBR) (Miller and Thode, 2007; Miller and Quayle, 2015). Next, the RAVG process calibrates the RdNBR values with the composite burn index (CBI; Key and Benson, 2006; Miller and Thode, 2007; Miller et al. 2009; Miller and Quayle, 2015). The CBI values are based on field plots where observed fire caused changes to vegetation strata (i.e., trees, shrubs, grasses, ground fuels) are quantified and ranked from 0 to 3 (Key and Benson, 2006) and this range of CBI values is used to classify landscape burn area by severity of burning: 0 equals no burning, 1 equals low severity, 2 moderate severity, and 3 represents high severity burn areas (Miller and Quayle, 2015). Then, RAVG maps the calibrated CBI (30 m pixels) into the three categories of low, moderate, and high to estimate fire severity across the burned landscape. We combined the two high severity classifications from RAVG into a single fire severity layer that displayed where tree cover loss due to urban fire in our three study areas was likely.

2.4. Urban neighborhood tree cover (nUTC) loss due to fire

The CFO 10 m nUTC data from pre- to post-fire years were extracted in the three pairs of fire and non-fire affected urban neighborhoods in the Thomas, Tubbs, and Camp fire perimeters. The area of nUTC loss were estimated by multiplying the sum of the 10 m resolution pixels by 100 to calculate square meters which were then converted to hectares. The percentage of nUTC loss was calculated as loss area from pre-fire to post-fire or between post-fire years divided by the pre-fire nUTC area in hectares. In addition, pre-fire and post-fire, nUTC percentage point change and percent loss were also estimated for five years post-fire to better understand the temporal nature of post-fire UTC dynamics. Accordingly, we estimated UTC change in both fire-affected (FA) neighborhoods and adjacent non-fire affected (NFA) neighborhood pairs in the three study areas. The sampling was done in comparable and equally sized areas in both FA and immediately adjacent NFA portions of the neighborhoods in Paradise (389 ha), Santa Rosa (940), and Ventura (254 ha).

We assessed accuracy in the nUTC loss estimates from pre- to post-fire years as well using visual interpretation of high-resolution Google Earth images based on Congalton's (2001) approach. Since the nUTC in the three areas of interest was less than 10 %, 250 random samples were used to assess the accuracy of nUTC loss due to fire in the high fire severity affected urban neighborhoods. All the Google Earth validation points were interpreted as reference data were then compared to the CFO 10 m nUTC loss in the form of an error matrix and accuracy estimates were reported for nUTC loss in three areas of interest during the analysis period (Kaspar et al., 2017). We also determined the amount of urban fire-affected neighborhood area located outside FRAP established WUI Interface and intermix areas. Finally, we used the McNemar Significance Test with the *mcnemar.test* function in R version 3.6.2 (R Core Team, 2021) to determine year to year statistically significant nUTC loss ($p < .05$) within the Thomas, Tubbs, and Camp fire perimeters.

2.5. Urban forest ecosystem service losses

We used existing information related to the ecosystem services provided by California's urban forests from McPherson et al. (2017) and developed normalized biome-specific proxies of per unit area tree cover-related ecosystem services (Table 1). The proxies were then used to estimate the amount of regulating ecosystem services being provided by existing forest cover in the three urban areas of study. These same values were then used to estimate per unit area nUTC loss of ecosystem services, specifically: carbon storage and sequestration, air pollution removal, energy savings, and stormwater interception (Table 1). We then used fire affected urban area and calculated area of nUTC that was in high and moderate severity to estimate ecosystem services loss; assuming that ecosystem service losses were exclusively in areas with greater burn severity impacts. Finally, we multiplied the area (m^2) of moderate and high severity with the proxy values in Table 1 to loss of ecosystem services, according to biome-specific ecosystem service values, for the three communities. We are not able to assess uncertainty in these proxy vales, however we do report in Table 1 standardized per unit of tree cover minimum and maximum estimates for air pollution removal from Nowak et al. (2006) and carbon storage and sequestration minimum estimates from Nowak et al. (2013).

3. Results

3.1. Neighborhood level urban tree cover change and trajectories

We estimated fire-affected nUTC in high and moderate fire severity areas in the three neighborhoods and tree canopy loss was expressed as both absolute change in area (hectares) as well as percentage point change (%). Specifically, Appendix A presents Urban Tree Cover (UTC) and Urban Neighborhood Tree Cover (nUTC) loss in both hectares and percentage point change from pre- to post-fire years due to wildfire. As expected, the majority of nUTC loss occurred from 2017 to 2018 in Ventura and Santa Rosa neighborhoods immediately following Thomas and Tubbs fire, respectively (Fig. 2). For example, nUTC loss in the 2017 Tubbs fire was 29 % from 2017 to 2018 (post-fire) but 19 % from 2016 to 2017 (pre-fire). Likewise, the 2017 Thomas fire and 2018 Camp fire had a nUTC of 44 % and 12 % of nUTC loss from 2017 to 2018 and 2018 to 2019, respectively. Overall, CFO data for Ventura and Santa Rosa show sharp decreases in the year or two following the fire but within 2 years their tree cover had increased to tree cover percentage and trends

Table 1
Per unit area urban tree cover-related ecosystem service proxies for Ventura, Santa Rosa and Paradise California, USA. Note: kg, kilograms; g, grams; m^2, square meters of urban tree cover. Minimum C (carbon) storage and sequestration values are from Nowak et al. (2013). Minimum air pollution removal values are from Nowak et al. (2006).

City	C storage (kg/m^2) [Minimum*]	C sequestration (kg/m^2/year) [Minimum*]	Air pollution removal (g/m^2/year) [Minimum-Maximum **]	Energy (kwh/year /m^2)	Stormwater (cm^3/year /m^2)	Source publication, (Biome)
Ventura	28.37 [4.59]	1.92 [0.017]	9.94 [5.5–28]	3.26	2.33×10^4	McPherson et al. 2017 (Southern California Coast)
Santa Rosa	39.92 [7.82]	3.25 [0.050]	3.44 [3.5–15.9]	0.91	7.17×10^4	McPherson et al. 2017 (Northern California Coast)
Paradise	27.26 [9.18]	1.82 [0.064]	7.09 [2.9–15.3]	0.38	5.76×10^4	McPherson et al. 2017 (Interior West)

* Minimum values standardized per unit tree cover are from Nowak et al. (2013). Los Angeles values were used for Ventura, Sacramento for Paradise, and San Francisco for Santa Rosa

** Includes: Particulate matter < 10 microns, Ozone, Sulphur dioxide and Nitrogen dioxide. Minimum and maximum values are standardized per unit tree cover and are from Nowak et al. (2006). Air pollution is the sum of: Particulate matter < 10 microns, Ozone, Sulphur dioxide and Nitrogen dioxide. Los Angeles values were used for Ventura, Sacramento for Paradise, and San Francisco for Santa Rosa

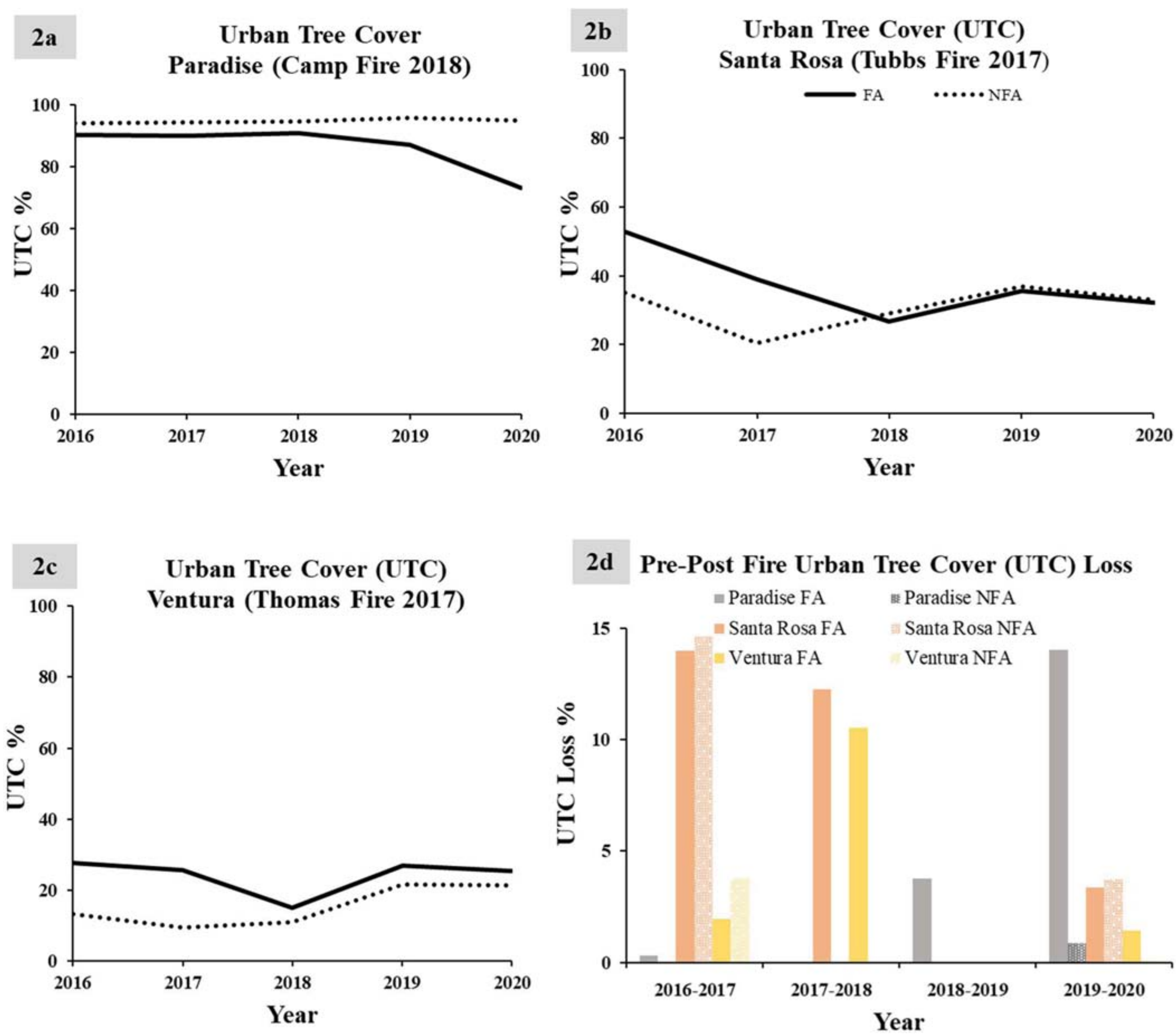

Fig. 2. Four-year neighborhood-level Urban Tree Cover (UTC) percentage point change in Fire-affected (FA) and adjacent Non-fire affected (NFA) areas in Paradise (2a), Santa Rosa (2b), and Ventura (2c), as well as percent UTC loss (2d) in the three study areas, USA using California Forest Observatory (2020) data.

similar to adjacent neighborhoods not affected by fire. By contrast, Paradise has been consistently and increasingly losing tree cover following the Camp fire (Fig. 2). For comparative purposes, Appendix B shows temporal urban tree cover changes using different imagery from the National Land Cover Database (See: https://www.usgs.gov/centers/eros/science/national-land-cover-database).

We also found that for each urban area, the year-to-year nUTC loss over the 3–4 post- fire period were all significantly different ($p < .05$) based on McNemar Significance Test (Table 2). The nUTC loss accuracy was also estimated from time of fire occurrence to post-fire years in the three neighborhoods and assessments are presented in Appendix (C, C1-C3; Congalton, 2001). Overall, we found that for Ventura, Santa Rosa, and Paradise, 21 %, 26 %, and 2 %, respectively, of the fire perimeter and affected urban area was outside the established WUI boundaries.

3.2. *Fire effects on urban forest ecosystem services*

Burn severity results show that of the 345 ha of the Ventura Thomas fire affected neighborhoods, 19 %, 39 %, and 36 % were in high, moderate, and low severity areas (Table 3). For Santa Rosa, 2560 ha were affected by the Tubbs fire and of these 8 %, 27 % and 19 % were in high, moderate, and low fire severity areas, respectively. In Paradise, the Camp fire affected 4679 ha and of these 8 %, 45 % and 39 % were in high, moderate, and low fire severity areas, respectively. The remaining proportions in the three neighborhoods consisted of unburned patches (Fig. 1). Ecosystem service losses were proportional to the area of nUTC lost due to fire. Paradise, with a larger urban area affected by fire and greater high and moderate burn severity areas (Table 3), had the greatest ecosystem service losses of all three study areas (Table 4). Conversely, Ventura had the least amount of ecosystem services loss.

4. Discussion

Global urban tree cover is about 25 % and these ecosystems are home to about 55 % of the world's human populations (Nowak and Greenfield, 2020). As such, the negative impacts of urban fires on cities and towns can result in a loss of ecosystem services for a large population of people in an affected area (Calkin et al., 2023). Additionally, climate change predictions suggest that by the end of the 21st century, we will see increased global surface temperatures of 1.5°C to 2°C along with an increased frequency and intensity of natural disasters such as wildfires worldwide (IPCC, 2021; Wasserman and Mueller, 2023; Underwood et al., 2019). These events are expected to have a detrimental impact on urban forest cover as well (Nitschke et al., 2017). Beyond the potential for future environmental adversity, the urban landscape already presents many challenges for urban forest ecosystems. Thus, more focus needs to be paid to mitigating these extreme urban fire events and resorting affected urban forest resources in these communities.

4.1. *Urban fire impacts to tree cover and ecosystem services*

It is important to emphasize that both urban tree cover and nUTC do not consist exclusively of planted trees and shrubs in private residential and commercial areas and public spaces such as institutional areas and parks (McPherson et al., 2017). Rather, a significant contribution to urban forest cover is from naturally regenerated or remnant trees and tall shrubs present in non-urban land covers across private parcels and other public spaces (e.g., parks, conservation areas, private undeveloped lots; Dobbs et al., 2011). Similarly, many of the affected standing trees survived following the fires and were able regenerate their crown during the analysis period (Gilbert and Brack, 2007). In terms of WUI versus urban fire affected areas, we found that in Ventura, Santa Rosa, and Paradise, 21 %, 26 %, and 2 %, respectively, of the fire affected neighborhoods were in urban residential and commercial land use covers outside the established WUI boundaries. Our analysis in moderate and high fire severity urban neighborhood in three fire perimeters indicated a nUTC loss of 44 % (39 ha) and 29 % (179 ha) in Thomas and Tubbs fire perimeters were observed following the fires (Appendix A).

Results from the Camp Fire indicate that nUTC loss can be significantly greater, 2 or 3 years after the fire (Tables 2 and 3). The reason for this delay in nUTC loss in Paradise could be due to the predominance of naturally growing, non-planted, forest dominated tree cover (Wang et al., 2022). More specifically, at nearly 600 m above sea level at the convergence between Sierran mixed hardwood and mixed conifer woodlands, vast tracts of the pre-fire urban forest in Paradise consisted of large trees and fuel loads that contributed to the catastrophic nature of the Camp Fire (Knapp et al., 2021). In residential areas, the pre-fire urban forest was likely dense, with a contiguous urban tree cover, with homes underneath (Fig. 2a; McNamara and Mell, 2022). Due to the slow recovery rate of this type of forest post-fire, often taking decades (van Wagtendonk et al., 2018), and the very high initial percent nUTC (Figs. 2a-2c), are likely factors behind Paradise's nUTC not showing any recovery during the analysis period. Further, while most dead and dying trees were removed soon after the fire, many more were removed in subsequent years due to safety concerns, poor tree health, tree mortality, rebuilding efforts, and infrastructure improvements (e.g., installation of underground electrical cables, transportation network improvements) leading to the current nUTC (Gilbert and Brack, 2007). The Town of Paradise lost approximately 80 % its original population and the reconstruction efforts have been slowed due to socio-political and economic issues (Troy et al., 2022), thus likely affecting the ability to replant trees both on private and public lands.

The more affluent city of Santa Rosa, located in a more sparsely vegetated, lower elevation oak woodland was affected by the Tubbs fire only in its far eastern neighborhoods (e.g., Coffey Park and Fountain Grove neighborhoods). The local community was able to quickly rebuild their homes and possibly livelihoods, while the rapidly resprouting and regenerating species trees (e.g., *Quercus spp.*) in both the public and private realms, contributed to the relatively rapid recovery of nUTC (Fig. 2b; McNamara and Mell, 2022). Of the original pre-fire tree species, coast redwoods, *Quercus spp.* and some palms species survived the Tubbs fire in several instances as observed by the authors. Field visits also showed that of the original 1000 street trees in Coffey Park, approximately 130 survived the fire and they were retained in the landscape largely following the residents' wishes to keep familiar elements in the post-fire disaster landscape.

The City of Ventura has the lowest precipitation and pre-fire tree cover of the three study areas (McPherson et al., 2017). It also had the least amount of WUI intermix areas since the fire affected neighborhoods were immediately adjacent to undeveloped areas with coastal

Table 2
Estimates of neighborhood-level urban tree cover (n UTC) loss in three selected fire perimeters due to wildfire using California Forest Observatory (CFO) 10 m data. Annual lost area in hectares was statistically and significantly different for all years and fires at $p < .05$*.

Fires	2016-2017	2017-2018	2018-2019	2019-2020
Thomas	18.7 %	44.4 %	10.1 %	22.2 %
Tubbs	18.6 %	29.0 %	6.7 %	13.9 %
Camp		3.2 %	12.0 %	17.3 %

Table 3
Urban Tree Cover loss was estimated in the high and moderate burn severity areas in the neighborhoods of three selected fire perimeters.

Urban Fire-affected neighborhoods	Urban Area (ha)	High Burn Severity (ha)	Moderate Burn Severity (ha)	Low Burn Severity (ha)
Thomas (Ventura)	345	66	136	125
Tubbs (Santa Rosa)	2560	216	684	479
Camp (Paradise)	4679	396	2108	1810

Table 4
Estimated total post-wildfire urban forest ecosystem service loss in Ventura, Santa Rosa and Paradise, California USA.

Fire (City)	Carbon Storage tons	Carbon sequestration tons/year	Air pollution removal* tons/year	Energy kwh/year	Stormwater m^3/year
Thomas (Ventura)	57,307	20,079	20	6591	47,066
Tubbs (Santa Rosa)	359,280	30,960	31	8187	645,300
Camp (Paradise)	682,590	177,534	178	9466	1442,304

* Includes: Particulate matter < 10 microns, Ozone, Sulphur dioxide and Nitrogen dioxide. Estimates are based on proxies obtained from McPherson et al. (2017).

sage vegetation in steep hill slopes with few buildings or structures. Pre-fire, there were visible areas of avocado and orange groves within the fire perimeter which also likely contributed to its overall tree cover (Figs. 2c and 2d). Like Santa Rosa, coastal Ventura was more affluent relative to Paradise and not only did nUTC rapidly recover to close to pre-fire levels (Fig. 2b), so did rebuilding and it also did not experience the population displacement of Paradise. Urban vegetation is also different from the other two cities in that palms, tall shrubs, and rapidly regenerating and resprouting trees like non-native *Eucalyptus spp*. and native *Platanus racemosa* and *Quercus spp*. are common.

Since urban fires are now a severe disturbance affecting communities, such events provide an opportunity to better understand the spatial and temporal fire effects on urban forests across a range of burn severities (Lentile et al., 2007) and to assess short-term socioecological impacts on cities and towns (Syphard et al., 2019; Yadav et al., 2023). Indeed, our UTC estimates can likely be used as an indicator of standing biomass (i.e., fuel loads) as measured by our remote sensing and RAVG methods (Tables 2 and 3) since they do show notable effects in low and moderate severity as well. Similarly, high severity fire in wildland areas has been reported to result in long-term changes to forests and subsequent negative impacts to ecosystems and communities (van Wagtendonk et al., 2018).

Although ecosystem service losses in the three cities (Table 4) were proportional to fire affected area and our tree cover change estimates (Table 3 and Fig. 2); the three different biome's ecological and geographic factors were also influential in changes in ecosystem service supply. Ventura has much lower average annual precipitation (43 cm) and tree cover than Santa Rosa (78 cm) and Paradise (147 cm). Thus, in terms of productivity (i.e., C storage and sequestration) as well as canopy interception and shading, (i.e., stormwater and energy) these biome difference will not only affect loss, but also the potential to recover these ecosystem services by restoring urban forest cover (Dobbs et al., 2011; Ossola et al., 2021). The UTC estimates on ecosystem service potential can also help to mitigate communities disproportionately burdened by wildfire risks in California (Yadav et al., 2023). Such impacts might indicate that urban fires will affect nUTC in the short term, but our finding show that in a few years UTC and ecosystem services supply can recover to pre-fire conditions (Fig. 2).

The ecosystem service change documented here has several significant implications for the use of urban forests as a nature-based solution. Considering that urban sprawl can increase fire occurrence and area burned, we expect that the urban and wildfire-driven losses may be even higher in the next few decades (Calkin et al., 2023; Wasserman and Mueller, 2023). But, given the amount of devastation in these post-fire communities that was described earlier in Section 2 (i.e., population displacement, housing burden, environmental pollution, socioeconomic impacts, community trauma), we were not able to estimate the socio-ecological impacts of ecosystem service loss to human wellbeing in these communities.

4.2. Urban versus wildland-urban interface wildfires as related to urban forests

Many studies discuss wildfire effects on ecosystems, buildings and structure loss, smoke emissions, and socioeconomic costs within wildland and WUI landscapes (Syphard et al., 2019; Troy et al., 2022; White and Zipperer, 2010). However, there is surprisingly little information from densely built, urbanized, and highly populated human settlements regarding their urban forests (Thomas et al., 2022). Although many of these communities and urban footprints are within WUI defined areas, the WUI concept was developed to address pre-fire management, planning, fire wise and home hardening fire risk reduction programs, and fire behavior modelling in ex-urban and suburban contexts with low density residential areas adjacent to ex-urban and rural areas (Radeloff et al., 2018; Stewart et al., 2009). Specifically, WUI is defined as either the 1) Intermix WUI, where structures and wildland vegetation intermingle, with both a housing density of > 6.2 houses/km^2 and >50 % of the area is continuous "wildland vegetation"; or 2) Interface WUI that are areas with <50 % vegetation and 2.4 km of continuous vegetated area, at least 75 % of which is "wildland vegetation", and that is at least 5 km^2 in size (Radeloff et al., 2018; Stewart et al., 2009).

As such, the application of a WUI lens by practitioners often focuses on more ex-urban, rural contexts and not high density residential, commercial, industrial land use-covers with greater human population densities (Schug et al., 2023; Radeloff et al., 2018). Similarly, the concept, to our knowledge, is not used for urban fires, post-fire restoration, planning, and management activities related to a community's urban forest cover (Bento-Gonçalves and Vieira, 2020; Jenerette et al., 2022). Although there has been wildfire related research from wildfires affecting WUI neighborhoods, this literature focuses on social dynamics and damage to buildings and infrastructure, but rarely in dense urban environments (Knapp et al., 2021; Thomas et al., 2022). Conversely, there is a lack of information on how fire can alter urban forest ecosystems, landscapes, and tree cover close to homes, commercial and industrial areas, and urban green spaces. As of 2018, approximately 55 % of the world's population live in urban areas and the health of billions of people is impacted by urban forests (de Guzman et al., 2022). Accordingly, we refer to these fire-affected systems as urban forests or the trees, woody plants, landscapes, and other vegetation in and around communities. Such information on fire-affected urban forests is necessary since urban areas are increasingly being affected by fires in many urbanized and urbanizing regions globally (UNEP, 2022).

This differentiation in terminologies between WUI and urban forests and their related practices and literature regarding fire-affected urban neighborhoods and communities is not trivial. At the community and neighborhood-level, many WUI-related fire mitigation practices and programs (e.g., fuels reduction, home hardening, fire-wise practices) specify the removal of most, if not all, trees, landscape vegetation, and organic material surface covers in areas adjacent to or near homes and buildings in order to reduce wildfire risk and hazard (https://www.nfpa.org/education-and-research/wildfire/preparing-homes-for-wildfire; Fig. 3). Conversely, urban forest and arboricultural best management practices and programs, recommend planting large, leafy trees, shrubs, and vegetation near homes (Fig. 3; Ball, 2011) to maximize ecosystem service benefits (e.g., increased property values, temperature reduction, carbon sequestration, particulate matter removal; Dobbs et al., 2011; McPherson et al., 2017). Although WUI fire mitigation plans and practices focus on the ecosystem disservices related to urban vegetation rather than their benefits; urban forest management plans on the other hand, rarely focus on costs (i.e., ecosystem disservices) such as increased fire risk from having more vegetation near homes (Escobedo et al., 2011; Gilbert and Brack, 2007).

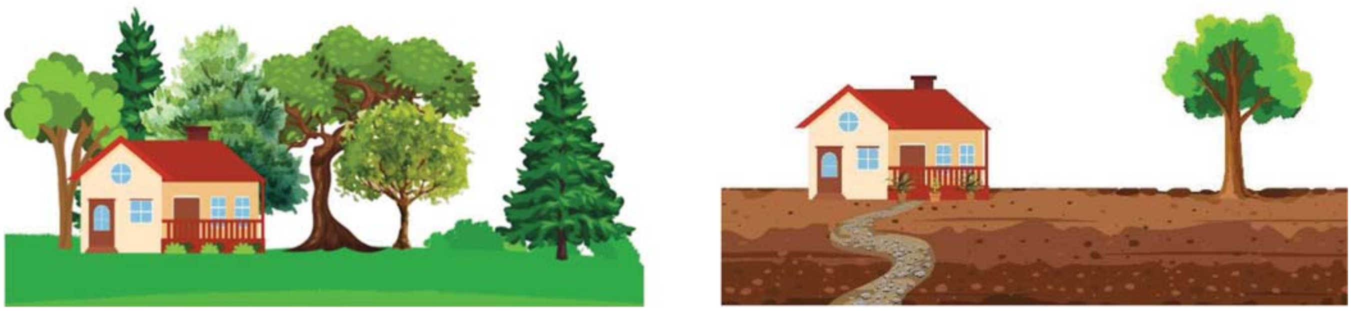

Fig. 3. Frequently recommended, but contradictory, urban forestry practices to maximize ecosystem service benefits to homeowners (left) versus Wildland-Urban Interface defensible space fire mitigation practices (right). Note the amount of vegetation and fuels immediately adjacent to the homes in both figures. Images courtesy of Adam Sanders, USDA Forest Service.

4.3. Implications for Urban forestry, arboriculture, restoring urban tree cover, and mitigating fire risk

The ecosystem services that urban trees provide have been found to mitigate the adverse effects of climate change (Dobbs et al., 2011; McPherson et al., 2017). Therefore, cities should plan for the impacts of climate change to maintain public health and safety (de Guzman et al., 2022; McDonald et al., 2020). As it relates to urban fires, steps need to be taken to protect not only people and property, but also trees. However, along with the benefits that urban trees provide come the potential risks and ecosystem disservices associated with their presence in densely populated areas (Escobedo et al., 2011). The issue is further complicated with the potential for an increased frequency and intensity of natural disasters associated with climate change. For example, following the 2003 brushfires in Canberra, Australia that damaged over 500 homes, there was a substantial increase in tree removal requests, 114 more than average, the following month (Gilbert and Brack, 2007). The authors suggest that this was partially due to the media's portrayal of natural disasters and how people then perceive such events to be associated with the trees and rather than the fire. Gilbert and Brack (2007) concluded that future urban forest management decisions need to consider public perceptions of trees as well as how people react to natural disasters. Furthermore, Ball (2011) explains that in the field of risk management, it is understood that perceptions of risk, media coverage of an incident, and the actual risk, are equally as problematic.

This dynamic further complicates the management decisions surrounding urban fires and treed landscapes in cities. The issue is multifaceted and needs to be addressed not only from an arboricultural perspective but also has impacts associated with educating the public on the benefits of trees and proactive maintenance to minimize potential risks (Ball, 2011). For example, WUI fire mitigation practices generally occur on larger tracts of land where mechanical treatments of vegetation and prescribed fire are used to mitigate wildfire risk (Fig. 3). These areas are also characterized by lower plant diversity and generally silvicultural methods are applied at the stand and landscape-level (van Wagtendonk et al., 2018). However, urban forestry and arboricultural practices mostly occur at the city-wide or in smaller, private parcels (Gilbert and Brack, 2007; Locke et al., 2022). These areas are highly diverse in terms of species composition and tree and shrub sizes (McPherson et al., 2017). Thus, individual tree-level, arboricultural practices are used that are often species and size-specific (Ossola and Hopton, 2018; Somerville et al., 2019). Additionally, urban trees are commonly subjected to compacted (Gregory et al., 2006) and polluted soils (Hagan et al., 2012), higher temperatures (Leuzinger et al., 2010), and experience frequent drought-like conditions (Nitschke et al., 2017). Such harsh growing environments have been linked to the high mortality rate of trees (Roman et al., 2014) in urban areas.

Despite the novelty of our study, it does have however, some limitations. First, we do not know the parcel-level effects of urban fires on certain tree-shrub species and their cover and the proportion of temporal tree loss in private versus public land tenure particularly in residential and commercial areas. This might be influencing the nUTC change dynamics seen in Figs. (2a-2c). Second, CFO and RAVG data are primarily used in wildland or WUI contexts; so our study is one of the first to apply these approaches to urban fires and urban land use/cover types. As such, we did not and could not readily, incorporate statistical measures of uncertainty for the RAVG severity metrics. Finally, our ecosystem service proxies are strictly supply-side estimates obtained from the literature (McPherson et al., 2017).

Adopting wildfire burn severity data and methods to urban land covers and fuel types could be used to further analyze the relationship of urban tree cover loss and fire severity. Indeed, adapting the use of high fire severity data (i.e., RAVG, advanced remote sensing techniques) to urban areas is warranted to better understand the spatiotemporal distribution of urban fire effects. Similarly, assessing whether the amount, type, condition, and moisture of vegetation near homes mitigates or facilitates building loss during wildfires, can have implications for the use of urban forests as natural climate solutions and protection of human life (Calkin et al., 2023). Future work is necessary to understand the extent and increasing occurrence of urban fire as a new disturbance agent in the cities of California and other urban areas globally (Thomas et al., 2022; Wasserman and Mueller, 2023). Over time, as the frequency and severity of urban fires increase, it is likely that the cost of damages and the adverse impacts to urban trees and communities will continue to rise (Escobedo et al., 2011; Yadav et al., 2023). Thus, research needs to address the potential solutions to mitigate the impacts of fire on urban forests.

Although there have been a few studies exploring the impact of fires on urban trees (Gilbert and Brack, 2007; Tessler et al., 2019), we still lack important pre-fire tree inventory data (e.g., health and condition, risk assessments, previous management inputs) as well as post-fire assessment data (i.e., severity of tree damage, likelihood of failure potential, alternative mitigation recommendations; Michael et al., 2018). Thus, urban forest data immediately before and after the fire event are needed to monitor and evaluate affected trees prior to removal as well as post-fire retained trees. Ideally, annual, long-term monitoring data could be used to better understand the impacts of fire on urban tree survival, performance, and potential risks. However, post-fire cleanup activities and human safety risks in these neighborhoods, makes it difficult for researchers to access these trees until months after a fire event. By this time, many damaged trees have already been removed.

That said, such information could inform proactive tree maintenance (i.e., structural pruning) that might aid in lessening the damage associated with these extreme events. Developing proper pruning techniques (e.g., single stem and lifting lateral branches high above the ground) is often referred to as removing ladder fuels in fire/fuels management and might reduce fire spread throughout the canopy of neighboring trees (i.e., crown fire). Likewise, creative landscape design solutions and proper selection of species or plant traits (e.g., deciduousness, moisture content) could help minimize flammability and damage to structures and canopy cover. That said, there is still a poor understanding of how tree health prior to a fire may affect a tree's susceptibility to fire damage (Wang et al., 2022). The effect of supplemental irrigation during the dry season might also have advantages when it comes to a tree's flammability and vulnerability to fire damage. Finally, a better understanding of plant flammability how different species perform during a fire could potentially help guide urban tree planting recommendations (White and Zipperer, 2010). This could limit the number of trees lost during a fire, help retain ecosystem service supply, while minimizing future fire risk. Similarly, planting new urban trees following a fire will not result in rapidly restoring tree cover, thus large mature trees need to be preserved so surrounding communities can maximize their associated ecosystem services.

5. Conclusion

Our current lack of understanding related to wildfire caused urban fires and effects on urban forests is concerning. Although the benefits of urban forests to society are well documented, we know very little when it comes to their protection during urban and WUI fire events. Given the frequency and severity of fires in recent years and the associated damage, research needs to work towards addressing solutions to limit the impacts of futures fires on our urban forests. A better understanding of potential ways to mitigate the negative effects of urban fires will help to promote and maintain public health and safety for urban populations across the globe.

As such, our findings contribute to a better understanding of urban forest trajectories following these emerging and extreme events. For example, it appears that Paradise, a forested ecosystem that was subsequently urbanized is better represented by the WUI concept. However, in afforested Ventura and urbanized Santa Rosa with a mix of remnant woodlands and afforested urban areas, formerly pastures, shrublands, and agricultural, affected areas were 20–25 % outside WUI defined areas. Therefore, although every community and urban forests will respond differently to urban fire impacts, this study highlights some commonalities and lessons. First, further studies on urban landscapes that have been previously affected by wildfires could quantify temporal changes and tree performance metrics. Second, existing tree cover datasets can help spatially predict where UTC is lost and gained could be used to monitor and evaluate the effectiveness of wildfire mitigation strategies. Indeed, the impacts of wildfire on UTC dynamics need to be explored in detail to manage threats to tree cover surrounding parcels in WUI areas. Third, UTC change analyses can be used to develop baseline comparative measures of tree cover loss during extreme fire events, specifically local scale burn severity analyses can document how urban fires affect long-term soil and water quality. Finally, ecosystem service assessments focused on resident's demand for benefits, perceived costs, or specific urban forest structure-fuel loading conditions could be used to promote the benefits that are lost post-fire and the potential benefits of tree plantings and restoration activities that algin with fuel mitigation practices.

Urban fires are becoming a novel threat to urban forests, thus differentiating WUI fuel treatments from urban forestry practices is more than a disciplinary, academic argument. Specifically, the application and trade-offs associated with these two approaches can possibly have implications for urban policymaker and homeowner decisions regarding home insurance coverage in high fire hazard areas. Similarly, traditional fuel treatment practices in urban areas can have the unintended consequence of increasing heat burden and reducing climate regulating services in disadvantaged communities with no air conditioning as well as stormwater regulation in high flood hazard areas, and impact real estate value premiums. Therefore, while WUI fire managers should be cognizant of the benefits that urban forests can provide to large segments of their communities; urban foresters should be aware of *urban and wildfire risk* and that urban landscape practices need to start accounting for potentially disastrous ecosystem disservices such as increased urban fire hazard that can result from increased tree cover and fuel loads in high-risk communities.

CRediT authorship contribution statement

Kamini Yadav: Writing – original draft, Visualization, Methodology, Investigation, Formal analysis. **Alessandro Ossola:** Writing – original draft. **Francisco J Escobedo:** Writing – original draft, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization. **Ryan Klein:** Writing – original draft. **Stacy Drury:** Writing – original draft, Resources, Methodology.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper

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Appendix A. Urban Tree Cover (UTC) and Neighborhood-level Urban Tree Cover (nUTC) loss area in hectares and percent from pre-fire to post-fire years. Bold values highlight years with noticeable losses

	Fire Name and Year of occurrence	CFO UTC (Ha) 2016	CFO UTC (Ha) 2017	CFO UTC (Ha) 2018	CFO UTC (Ha) 2019	CFO UTC (Ha) 2020	CFO UTC Loss Ha (%) 2016–2017	CFOUTC Loss Ha (%) 2017–2018	CFO UTC Loss Ha (%) 2018–2019	CFO UTC Loss Ha (%) 2019–2020
Urban Neighborhood Tree Cover (nUTC)	Thomas Fire Ventura (2017)		**86.76**	**48.47**	90.89		17.2 (18.67 %)	**38.52 (44.4 %)**	4.9 (10.10 %)	20.2 (22.22 %)
	Tubbs Fire Santa Rosa (2017)	818.99	**616.77**	**433.79**	567.47		151.98 (18.55 %)	**178.6 (28.96 %)**	29.01 (6.69 %)	78.62 (13.85 %)
	Camp Fire Paradise (2018)		3770.3	**3808.68**	**3406.30**	2707.77		118.63 (3.15 %)	**454.7 (11.94 %)**	587.49 (17.25 %)

Appendix B. Four-year Urban Tree Cover (UTC) percentage point change in Paradise, Santa Rosa, and Ventura CA in Fire (FA) and adjacent Non-fire affected (NFA) neighborhoods using National Land Cover Data

	Fire Name and Year of occurrence	CFO UTC (Ha) 2016	CFO UTC (Ha) 2017	CFO UTC (Ha) 2018	CFO UTC (Ha) 2019	CFO UTC (Ha) 2020	CFO UTC Loss Ha (%) 2016-2017	CFOUTC Loss Ha (%) 2017-2018	CFO UTC Loss Ha (%) 2018-2019	CFO UTC Loss Ha (%) 2019-2020
Urban Neigh-borhood Tree Cover (nUTC)	Thomas Fire Ventura (2017)		**86.76**	**48.47**	90.89		17.2 (18.67%)	**38.52 (44.4%)**	4.9 (10.10%)	20.2 (22.22%)
	Tubbs Fire Santa Rosa (2017)	818.99	**616.77**	**433.79**	567.47		151.98 (18.55%)	**178.6 (28.96%)**	29.01 (6.69%)	78.62 (13.85%)
	Camp Fire Paradise (2018)		3770.3	**3808.68**	**3406.30**	2707.77		118.63 (3.15%)	**454.7 (11.94%)**	587.49 (17.25%)

Appendix C. User and producer accuracy Assessment of neighborhood-level urban tree over (nUTC) loss in the Camp (Paradise), Tubbs (Santa Rosa), and Thomas (Ventura) Fire perimeters using California Forest Observatory data at 10 m spatial resolution. The user's accuracy of nUTC loss from fire occurrence year to post-fire year for the three selected fire perimeters were lower than no change class. The user's accuracy of the nUTC loss in the Camp fire perimeter from 2018 to 2019 was 17 % showing omission from the UTC loss. The user's accuracy of nUTC loss in Thomas and Tubbs fire perimeters were 55 % and 30 % from 2017 to 2018. The lower user's accuracy shows that the samples in the loss class from fire year to post-fire were not accurately mapped and were subsequently omitted from the map. Appendices C1, C2, and C3 display producer and user accuracy assessments for fire occurrence and post-fire years for the three different fire perimeters

	C1. Sampling Reference Data Thomas Fire (Ventura) 2017–2018					
		NC	Loss	Gain	Total	Users
CFO Map Data	NC	175	15	2	192	0.00 %
	Loss	40	18	0	58	31.03 %
	Gain	0	0	0	0	0.00 %
Total		215	33	2	250	
Producer		81.40 %	54.55 %	0.00 %		**77.20 %**
	C2.Sampling Reference Data Tubbs Fire (Santa Rosa) 2017–2018					
		NC	Loss	Gain	Total	User
CFO Map Data	NC	189	35	0	224	84.38 %
	Loss	9	15	0	24	62.50 %
	Gain	1	1	0	2	0.00 %
Total		199	51	0	250	
Producer		94.97 %	29.41 %	0.00 %		**81.60 %**
	C3. Sampling Reference Data Camp Fire (Paradise) 2018–2019					
		NC	Loss	Gain	Total	User
CFO Map Data	NC	90	108	0	198	45.45 %
	Loss	7	23	0	30	76.67 %
	Gain	15	7	0	22	0.00 %
Total		112	138	0	250	
Producer		80.36 %	16.67 %	0.00 %		**45.20 %**

CFO: California Forest Observatory 10 m data; NC: No Change; Gain: No tree to Tree; Loss: Tree to No-Tree;

Appendix C.1. . Camp Fire accuracy assessment

	Sampling Reference Data Camp Fire (Paradise) 2017-2018					
		NC	Loss	Gain	Total	User
Map Data	NC	222	3	2	227	97.80 %
	Loss	11	3	0	14	21.43 %
	Gain	7	1	1	9	11.11 %
Total		240	7	3	250	
Producer		92.50 %	42.86 %	33.33 %		**90.40 %**
	Sampling Reference Data Camp Fire (Paradise) 2019–2020					
		NC	Loss	Gain	Total	User
Map Data	NC	161	1	22	184	87.50 %
	Loss	49	0	5	54	0.00 %
	Gain	10	0	2	12	16.67 %
Total		220	1	29	250	
Producer		73.18 %	0.00 %	6.90 %		**65.20 %**

Appendix C.2. . Thomas Fire accuracy assessment

	Sampling Reference Data Thomas Fire (Ventura) 2016-2017					
		NC	Loss	Gain	Total	User
Map Data	NC	199	5	0	204	97.55 %
	Loss	24	0	0	24	0.00 %
	Gain	21	0	1	22	4.55 %
Total		244	5	1	250	
Producer		81.56 %	0.00 %	100.00 %		**80.00 %**
	Sampling Reference Data Thomas Fire (Ventura) 2018–2019					
		NC	Loss	Gain	Total	User
Map Data	NC	187	1	5	193	96.89 %
	Loss	0	0	0	0	0.00 %
	Gain	51	2	4	57	7.02 %
Total		238	3	9	250	
Producer		78.57 %	0.00 %	44.44 %		**76.40 %**

Appendix C.3. . Tubbs Fire accuracy assessment

	Sampling Reference Data Tubbs Fire (Santa Rosa) 2016-2017					
		NC	Loss	Gain	Total	User
Map Data	NC	205	0	1	206	99.51 %
	Loss	37	0	0	37	0.00 %
	Gain	6	1	0	7	0.00 %
Total		248	1	1	250	
Producer		82.66 %	0.00 %	0.00 %		**82.00 %**
	Sampling Reference Data Tubbs Fire (Santa Rosa) 2018–2019					
		NC	Loss	Gain	Total	User
Map Data	NC	222	1	2	225	98.67 %
	Loss	5	0	0	5	0.00 %
	Gain	17	0	3	20	15.00 %
Total		244	1	5	250	
Producer		90.98 %	0.00 %	0.00 %		**90.00 %**

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