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Title: The contribution of Indigenous stewardship to an historical mixed-severity fire regime in British Columbia, Canada

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Abstract

Indigenous land stewardship and mixed-severity fire regimes both promote landscape heterogeneity, and the relationship between them is an emerging area of research. In our study, we reconstructed the historical fire regime of *Ne Sextsine,* a 5900-ha dry, Douglas-fir-dominated forest in the traditional territory of the *T'exelc* (Williams Lake First Nation) in British Columbia, Canada. Between 1550 and 1982 CE, we found median fire intervals of 18 years at the plot-level and 4 years at the study site-level. *Ne Sextsine* was characterized by an historical mixed-severity fire regime, dominated by frequent, low-severity fires indicated by fire scars, with infrequent, mixed-severity fires indicated by cohorts. Differentiating low- from mixed-severity plots through time was key to understanding the drivers of the fire regime at *Ne Sextsine*. Low-severity plots were coincident with areas of highest use by the *T'exelc*, including winter village sites, summer fishing camps, and travel corridors. The high fire frequency in low-severity plots ceased in the 1870s, following the smallpox epidemic, the forced relocation of Indigenous peoples into small reserves, and the prohibition of Indigenous burning. In contrast, the mixed-severity plots were coincident with areas where forest resources, such as deer or certain berry species, were important. The high fire frequency in the mixed-severity plots continued until the 1920s when industrial-scale grazing and logging began, facilitated by the establishment of a nearby railway. *T'exelc* oral histories and archaeological evidence at *Ne Sextsine* speak to varied land stewardship, reflected in the spatiotemporal complexity of low- and mixed-severity fire plots. Across *Ne Sextsine*, 63% of cohorts established and persisted in the absence of fire after colonial impacts beginning in the 1860s, resulting in a dense, homogenous landscape that no longer supports *T'exelc* values and is more likely to burn at uncharacteristic high severities. This nuanced understanding of the Indigenous contribution to a mixed-severity fire regime is critical

for advancing proactive fire mitigation that is eco-culturally relevant and guided by Indigenous expertise.

Keywords: Mixed-severity fire regime; Indigenous fire stewardship; dendrochronology; British Columbia; dry forests; fire history; Douglas-fir; tree-rings

Introduction

People and fire have coexisted for millennia in fire-prone landscapes, and yet today there is widespread concern that fire is increasingly threatening the things people value (Bowman et al. 2009, 2011). Globally, climate change, population growth, and a legacy of fire exclusion through disruption of Indigenous stewardship and fire suppression policies are interacting to create a complex wildfire challenge that cannot be solved by a one-size-fits-all approach (Krebs et al. 2010, Bowman et al. 2011, Stephens et al. 2013). Disentangling the complexity of the modern wildfire challenge requires an historical perspective that recognizes the dynamic interactions among fire, people, and the land through time (Moritz et al. 2014, Spies et al. 2014, Fischer et al. 2016, Smith et al. 2016) while acknowledging and adapting to the realities of the present and future (Falk et al. 2019, Prichard et al. 2021). This historical perspective can help enable a continuum of resilience approaches that minimize the likelihood of negative consequences in the future and support a return to coexisting with (rather than controlling) fire (Moritz et al. 2014, Smith et al. 2016, McWethy et al. 2019, Prichard et al. 2021).

The fire regime concept characterizes patterns and processes of fire across multiple spatial and temporal scales (Krebs et al. 2010, Turner 2010, Whitlock et al. 2010). Fire regimes are often described along a continuum of two key characteristics - fire frequency and fire severity. At one end of the continuum fire regimes are dominated by high-frequency, low-severity fires causing low levels of plant mortality, while at the other end fire regimes are dominated by low-frequency, high-severity fires causing higher levels of plant mortality (Agee 1993, Schoennagel et al. 2004). In recent decades, mixed-severity fire regimes (MSFR) that contain some proportion of low- to high-severity fires (Daniels et al. 2017) are increasingly recognized as a primary type of fire regime in western North America (Amoroso et al. 2011, Perry et al. 2011, Heyerdahl et al. 2012, Marcoux et al. 2015, Chavardès and Daniels 2016, Hessburg et al. 2016, Brookes et al. 2021).

Characterizing fire regime drivers can be complex, in part because of the different scales at which drivers operate (Syphard et al. 2017). Fire regimes can be driven by top-down (e.g., climate) or bottom-up (e.g., topography, fuels, ignitions) factors (Agee 1993, Perry et al. 2011). Fire regimes that are driven by top-down factors tend towards low-frequency, high-severity fires whereas fire regimes that are driven by bottom-up factors tend towards high-frequency, low-severity fires. MSFR occur when top-down and bottom-up drivers interact, creating complex and heterogenous landscapes (Perry et al. 2011, Daniels et al. 2017). In the natural science literature, climate, topography, and fuels drivers are well documented (Schoennagel et al. 2004). In contrast, Indigenous fire stewardship as a driver of fire regimes is not well represented, in part because of an assumption that Indigenous stewardship did not influence landscape-level fire patterns (Leonard et al. 2020, Roos 2020, Hoffman et al. 2022). Failure to consider the contribution of Indigenous stewardship to fire regimes at different scales may overlook the importance of ignition sources as a bottom-up fire regime driver (Krebs et al. 2010) and result in interpretations and management recommendations that favor priorities of non-Indigenous peoples (Prichard et al. 2021, Copes-Gerbitz et al. 2022b, Hoffman et al. 2022).

Ethnographic research, oral histories and contemporary fire knowledge keepers describe Indigenous fire use for multiple objectives at a variety of scales (Huffman 2013, Lake and Christianson 2019). In western North America, Indigenous fire stewardship – including intentional ignitions and stewarding fire-affected landscapes (Lake and Christianson 2019, Hoffman et al. 2022) – achieves a variety of cultural and ecological objectives, including increasing abundance of preferred resources, promoting desired landscape conditions, and fulfilling an obligation to respect the land (Turner et al. 2000, Kimmerer and Lake 2001, Huffman 2013, Miller and Davidson-Hunt 2013, Trauernicht et al. 2015, Lewis et al. 2018, Lake and Christianson 2019). These objectives are spatiotemporally heterogeneous, and often include a higher incidence of intentional ignitions during appropriate seasonality near communities and along travel corridors that connected highly-valued resource areas (Turner et al. 2003, Lake and Christianson 2019, Roos et al. 2021). While Indigenous fire stewardship is a year-round practice, intentional ignitions tend to occur in the shoulder seasons (spring and autumn), when fire risk is low and primary objectives of burning can be met (Nikolakis et al. 2020, Hoffman et al. 2022). The spatiotemporal heterogeneity of Indigenous fire stewardship is a unique pyrodiversity that has both the intentional objective and added benefit of reducing the likelihood of future detrimental fires (Christianson 2015, Trauernicht et al. 2015, Bird et al. 2016, Mistry et al. 2016, Taylor et al. 2016, Hoffman et al. 2021, Mariani et al. 2021). Despite severe limitations on the continuation of these practices because of the impacts of colonization, Indigenous fire stewardship (including knowledge and practice) continues to adapt to modern social, ecological and climatological contexts (Eriksen and Hankins 2014, Lewis et al. 2018, Lake and Christianson 2019). In places where this fire stewardship was disrupted or lost, it follows that the resulting homogeneous landscapes no longer support Indigenous livelihoods nor maintain the benefit of minimizing the impact of future fires (Hoffman et al. 2021, Mariani et al. 2021).

This modern landscape homogenization also results when MSFRs are altered, leading to simplified forest structure and composition that increase the likelihood of uncharacteristic, large, high-severity fires (Schoennagel et al. 2004, Stephens et al. 2013, Marcoux et al. 2015, Chavardès and Daniels 2016, Hessburg et al. 2019, Hagmann et al. 2021). MSFRs are characterized by pyrodiversity that drives patch- to landscape-scale diversity (Hessburg et al.

2016), which parallels the local- to landscape-scale objectives of fire use by Indigenous peoples (Kimmerer and Lake 2001, Lake and Christianson 2019). Understanding the contribution of Indigenous stewardship to fire regimes is critical because characterizations of historical fire regimes help drive and inform management approaches, especially those that seek to reintroduce fire (White et al. 2011, Hessburg et al. 2016). Understanding the influence of Indigenous ignitions on fire regimes is an emerging area of historical fire ecology through dendroecological (Fulé et al. 2011, Stambaugh et al. 2013, Taylor et al. 2016, Guiterman et al. 2019, Larson et al. 2020, Kipfmueller et al. 2021) and palaeoecological methods (McWethy et al. 2013, Roos et al. 2014, 2021). Nevertheless, for lessons from historical ecology to continue to be relevant in a management context - such as ecosystem management based on historical fire regimes - it must embrace emerging trends (Swetnam et al. 1999, Higgs et al. 2014), such as acknowledging the devastating impacts of colonialism on Indigenous fire stewardship and a commitment to learn from and work with Indigenous communities in addressing the modern risk of fire (Lake et al. 2017, Lake and Christianson 2019, Dickson-Hoyle et al. 2021).

In the dry forest ecosystems of British Columbia (BC), Canada, modern wildfires have negatively impacted communities and their livelihoods, with seven significant fire seasons affecting the wildland-urban interface since 2000. The 2017 fire season, as an example, prompted the evacuation of over 65,000 people, burned multiple homes and over 1.2 million hectares, and initiated a 70-day provincial state of emergency (Abbott and Chapman 2018). The fires in 2017 were consistent with a pattern of increasing area burned, number of large fires (>200 ha), and lightning-caused fires, and a lengthening of the fire season in the southern Canadian cordillera (Hanes et al. 2019) and were exacerbated by anthropogenic climate change (Kirchmeier-Young et al. 2019). Furthermore, the 2017 fire season forced the evacuation of 26

First Nations communities in BC alone (Abbott and Chapman 2018), reflecting the disproportionate impacts on Indigenous peoples across Canada (Christianson 2015, Asfaw et al. 2019, McGee et al. 2019, Zahara 2020, Erni et al. 2021). The Constitution of Canada recognizes three distinct groups of Indigenous Peoples: First Nations, Métis, and Inuit. We use Indigenous throughout, except where specific references or names indicate First Nation(s) is more appropriate. Wildfires of this magnitude will continue to challenge future suppression efforts (Wotton et al. 2017) and drastically increase expenditures (Stocks and Martell 2016) without significant interventions to address existing risk (Johnston et al. 2020). In BC, the provincial government is the primary decision-maker for fire management that has historically been fire suppression-focused (Nikolakis and Roberts 2021, Copes-Gerbitz et al. 2022b), although ongoing challenges from recent wildfire seasons are highlighting the need to incorporate the expertise of Indigenous and local communities to guide proactive approaches to management (Dickson-Hoyle and John 2021, Copes -Gerbitz et al. 2022a).

In landscapes with overlapping governance and values, such as those where Indigenous traditional territories and the wildland-urban interface overlap, proactive management must reflect both the ecological and cultural context (Blum 2004, Copes-Gerbitz et al. 2022b). This context includes an understanding of how the relationship between fire, people, and the land has changed through time. Thus, our study sought to: 1) characterize the historical fire regime, including fire frequency and fire severity, and 2) explore the contribution of Indigenous fire stewardship to the fire regime. We focused this study in a Community Forest that is within the *Secwepemcúl'ecw* – the traditional territory of the Secwépemc peoples – where our ongoing collaborative research provides a foundation for supporting Indigenous-led approaches to addressing wildfire risk (Copes-Gerbitz et al. 2021, 2022c). In keeping with recommendations for natural scientists working with Indigenous communities, we centered Indigenous knowledge in interpreting the fire history story rather than the problematic approach of integrating only "relevant" knowledge into Western natural science frameworks, which can be considered an ongoing form of colonization (Bohensky and Maru 2011, Mistry and Berardi 2016, Wong et al. 2020).

Methods

Study area

Ne Sextsine is a ~5,900-hectare (ha) area that is part of the broader territory of the *T'exelc* (Williams Lake First Nation), who have continuously stewarded the area since time immemorial. In 2014, *Ne Sextsine* was incorporated into the Williams Lake Community Forest (WLCF) and is currently co-managed by the *T'exelcemc* (the people of the Williams Lake First Nation) and the City of Williams Lake for a broad range of social, cultural, ecological, and economic values (WL Community Forest LP 2015). *Ne Sextsine* is located in the wildland-urban interface of the City of the Williams Lake (population ~12,000) and the Cariboo-Chilcotin Natural Resource District in south central BC (**Figure 1**). *Ne Sextsine* was not directly affected by fires in 2017, but ~12% (870,000 ha) of the Cariboo-Chilcotin Natural Resource District was burned and the communities in the wildland-urban interface of *Ne Sextsine* were evacuated (Government of BC, 2017). As such, community protection from wildfire is a key mandate driving land-management decisions at *Ne Sextsine*.

Ne Sextsine is bounded by steep river valley cliffs to the west and north, with gently rolling topography to private and municipal land to the south and east. The elevation ranges from 400–1000 m above sea level (m.a.s.l.). Climate is continental and strongly controlled by the rain shadow of the Coast Mountains; average monthly temperature ranges from -7°C in December to 16°C in July (Environment Canada, 2020). Total annual precipitation is 451 mm, with highest rainfall in June (59 mm) and highest snowfall in December (45 cm) (Environment Canada, 2020). Based on fire-weather data from 2005–2019, 17% of days during the calendar year on average were rated as high to extreme fire danger (typically July-August); in 2017, 30% of days were rated as such (Government of BC, 2021).

The landscape at *Ne Sextsine* is dominated by interior Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) forests, including a drier forest type (~2400 ha) primarily along the benches of the Fraser River and more mesic forest type (~3500 ha) on a higher elevation plateau, with about 100 ha of existing grassland that gradually transitions to forest (WL Community Forest LP 2015). Aerial photos from the 1960s-70s show grasslands at *Ne Sextsine* covered ~340 ha, which is mandated for grassland restoration as part of the Grasslands Strategy of the Cariboo-Chilcotin Land Use Plan (Steele et al. 2007). The mesic forest includes trembling aspen (*Populus tremuloides*) and hybrid spruce (*Picea engelmannii* x *glauca*) in the wetter areas (WL Community Forest LP 2015). It also includes a small living component of lodgepole pine (*Pinus contorta*), although mature lodgepole is largely absent in the canopy due to the mountain pine beetle epidemic in the early 2000s (WL Community Forest LP 2015).

Field methods

Based on early conversations with the Williams Lake First Nation archaeologist and Elders, we aimed to capture the full range of known archaeological and ecological variation at *Ne Sextsine* (Copes-Gerbitz et al. 2021, 2022c). To do so, we used a stratified-systematic sampling design (**Figure 1**) to differentiate the dry forest benches (hereafter, "bench") from the mesic forest (hereafter, "plateau"). We placed a 0.5 km grid over the study site and selected alternate vertices on the bench (plots 1 km apart) and every fourth vertex on the plateau (plots 2 km apart). Of the 59 potential sample plots, five were excluded because they were not safely accessible. At the 54 accessible plots (n=34 and 20 on the bench and plateau, respectively), location, elevation (m.a.s.l.), slope angle and aspect were recorded. Where plots landed within a recorded archaeological site or where archaeological evidence was identified in the field, the plot location was moved on a random bearing up to 100 m away so as not to disturb cultural sites.

We used a modified n-tree design (Jonsson et al., 1992; Lessard et al., 2002) to sample the 20 living trees or snags (snag stage ≤4; Thomas et al., 1979) closest to plot center, including 10 canopy and 10 subcanopy individuals with diameter at breast height (DBH) ≥12.5 cm. For each individual, we recorded species, DBH, height class (canopy or subcanopy), status (live or dead), and snag decay class (Thomas et al. 1979). To estimate age, one increment core was collected as close to the pith (center) as possible and near the root collar, and coring height was recorded (Daniels et al. 2017). The distances from plot center to the outermost tree or sound snag (decay class ≤4) in the canopy and subcanopy classes were recorded to calculate scaling factors for circular plots to convert individual trees to a number of trees ha^{-1}, which we used to calculate tree density (Lessard et al. 2002). Stumps and decayed snags (decay class ≥5) were counted in the larger of the two canopy or subcanopy circular subplots. Saplings were tallied by species in a 3.99m radius subplot (0.005 ha) around plot center.

In a ~1 ha circular area (radius = 56.4 m) around each plot center, we searched for fire-scarred trees, snags, logs, and stumps (hereafter "scarred trees") and noted potential culturally modified trees. We were trained to recognize archaeologically significant and culturally modified trees by the archaeologist for the Williams Lake First Nation (Archaeology Branch 2001, Copes-Gerbitz et al. 2022c). From the 37 plots with fire evidence, we selected 20 plots (10 on the bench and 10 on the plateau) with at least five scarred trees with multiple visible scars of potentially varying ages. We prioritized snags, logs and stumps for chainsaw sampling cross-sections, took partial sections on living trees where safe to do so (Cochrane and Daniels 2008), and avoided all trees of archaeological significance that are protected by law in BC (Archaeology Branch 2001).

We also avoided sampling trees within archaeological sites and in areas of cultural significance as requested by the Williams Lake First Nation (Copes-Gerbitz et al. 2022c). Plots where fire scars were collected (n = 20) are hereafter "fire plots" (that contain both age data and fire scar data) where plots with no fire scars (n = 34) collected are hereafter "age plots" (and contain just age data).

Dendrochronological analyses

Increment cores and fire-scar samples were mounted on wooden supports and sanded to reveal ring structure (Stokes and Smiley 1996). Samples were scanned at high resolution (2400 dots per inch) and measured using the program CooRecorder (Larsson 2011*a*). Ring widths were visually and statistically dated against a master chronology for the region (Daniels and Watson 2003) using the programs COFECHA and Cdendro (Holmes 1983, Larsson 2011*b*); seven samples (one increment core and six fire scars) could not be accurately crossdated and were not used in further analyses. We recorded inner- and outer-most ring dates for each sample (Swetnam and Baisan 1996). Tree ages were estimated by applying (1) a coring-height correction and (2) a pith correction where tree cores missed the pith (Duncan 1989). The number of years for trees to grow to coring height was estimated using species-specific age-on-height linear regression models for *P. menziesii* (correction = 0.54 (coring height in cm) / 6.87) and *P. contorta* (correction = 0.30 (coring height in cm) / 5.07). These models were originally developed by sectioning seedlings at 10-cm intervals to estimate seedling height-growth rates, then applying adjustments for radial growth rates classified based on ring widths (Daniels and Watson 2003). The *P. menziesii* model was also used for *Picea* sp. since both species are moderately shadetolerant. Cores with total age corrections ≤25 years were retained (n=1049) and 92% of age corrections were <15 years, so age structures were presented in 15-year classes.

Year and seasonality (earlywood, latewood, dormant, unknown) for each fire scar was noted by identifying the position of the scar tip compared to the timing of ring formation using a microscope (Swetnam and Baisan 1996). Seasonality can be an important indicator for Indigenous fire stewardship found in tree-rings (Granström and Niklasson 2008, Kipfmueller et al. 2021). We sampled tree cores between May and October, and observed ring formation from late May (earlywood) to early August (latewood). Although the timing of ring formation is related to intra- and inter-annual variation in moisture deficits (Hope et al. 1991), we interpreted seasonality of scars as spring (earlywood), mid-summer (latewood), or late-summer or autumn (dormant season between two rings). Dormant season scars can form at any time between the end of the growing season of the year of ring formation to the start of the growing season of the subsequent year (Dieterich and Swetnam 1984), so we assigned calendar years to dormant season scars based on the following criteria: (1) the year of ring formation if a latewood scar was present in another tree in the same or a proximal plot, (2) the subsequent year if an earlywood scar was present in another tree in the same or a proximal plot, or (3) otherwise, the year of ring formation given that the modern peak in fire activity is late summer in this forest type.

Reconstructing fire frequency and severity

We compiled fire chronologies for individual fire plots, and composite fire chronologies for fire plots located on the bench, plateau and entire study area using FHAES software (v 2.0.2). For each chronology, we calculated the length of the fire record (first to last scar), and the number, Weibull median, and range of scar-to-scar intervals (Brewer et al. 2020). Fires that scarred trees in ≥30% recording plots were considered widespread within our 5,900 ha study area. We chose ≥30% recording plots because it best captured visible peaks of fire activity after testing three different criteria (≥50%, ≥30%, and ≥20%) and because it was similar to the criteria in nearby studies in dry forest ecosystems (≥25%; Harvey et al. 2017, Brookes et al. 2021). Fire severity through time was determined at the plot-scale (in fire plots and age plots) based on the presence of fire scars and even-aged cohorts (Heyerdahl et al. 2012).

Cohorts were identified as a 15-year period in which ≥25% of trees ha^{-1} established in the plot (Marcoux et al. 2015). We chose a 15-year period because it was less than the median plot-level fire return interval, allowing us to assess if cohorts were initiated by fire. Starting from the oldest pith date in the plot, we tested successive pith dates to determine the start of the cohort, which was defined by the oldest pith date from which ≥25% of trees ha^{-1} established in a 15-year window. We continued testing whether subsequent pith dates met the cohort criteria; the end date of the cohort was the last pith date that contributed to the ≥25% of trees ha^{-1} criteria in the last 15-yr window. We tested whether excluding cohorts with some tree establishment in the 15 years immediately preceding the cohort start year was warranted (Heyerdahl et al. 2012); however, this criterion did not detect visible pulses of establishment and, thus, was not used in our analyses.

To determine whether cohorts were initiated by fire, we examined multiple lines of evidence. Cohorts were classified as fire-initiated if the cohort start date was within 15-years after or 5-years before a fire-scar date (to account for age correction errors). Because fire scars were only collected at 20 fire plots, fire-scar dates included those within a fire plot or in an adjacent fire plot up to 2km away. Cohorts that did not meet these criteria were deemed 'unattributed'; these unattributed cohorts were not considered a result of abiotic (windthrow, freezing events) or biotic (insects and pathogens) disturbances because we did not observe groups of dead or down canopy trees in any plots. Fire-initiated and unattributed cohorts were further classified based on timing and persistence. First, we determined whether cohorts established prior to or after the most recent widespread fire (1870), given that post-1870s cohorts may be a legacy of decreasing fire frequency. Second, cohorts were considered persistent if their end date was after the last fire scar in individual fire plots or in ≥1 fire plot adjacent to age plots.

Finally, we combined multiple lines of evidence to determine fire severity through time at the plot scale. We focus on fire severity through time rather than event-level fire severity following the methods applied in other dry conifer forests of BC (Heyerdahl et al. 2012, Harvey et al. 2017) and given the methodological challenges with reconstructing event-level fire severity through time (Daniels et al. 2017). All fire and age plots had cohorts; therefore, plots were assigned either low-, mixed-, or high-severity fire history through time depending on the cohort attribution (fire-initiated or unattributed), timing and persistence. Fire and age plots were assigned either: (1) low-severity if all cohorts (regardless of attribution) persisted in the absence of subsequent fire; (2) mixed-severity if ≥1 cohorts (regardless of attribution) established prior to1870 and survived subsequent fires; or (3) high-severity if an unattributed cohort established prior to 1870 and no sampled trees were alive prior to cohort establishment. Plots that could not be assigned a severity were undetermined.

Indigenous contribution to the fire regime

Through ongoing collaborative work with the *T'exelc*, we explored the spatial and temporal differences in the dendroecological fire regime in the context of place-based oral histories and ethnographic records (Copes-Gerbitz et al. 2021, 2022c). Spatially, we considered whether differences in fire regime characteristics (e.g., plot-level fire severity through time, seasonality) were aligned with different stewardship practices and use of *Ne Sextsine*, including winter village sites, summer fishing camps, travel corridors, or areas where berry-picking and hunting were common (Copes-Gerbitz et al. 2021). Given that winter village sites and summer fishing camps were more likely to be located along the benches compared to the plateau, we also tested for spatial differences in physical plot attributes between the bench and plateau using Mann-Whitney U tests (Mann and Whitney 1947), except for aspect which was compared using a t-test after transformation to linear scale ranging from 0 (45°) to 180 (225°) along a cool to warm gradient. Temporally, we tested for changes in cumulative fire frequency using piecewise linear regression in SigmaPlot (v 13.0) and compared the break points to key dates of land use change, following methods used in similar fire history studies (Guiterman et al. 2019). We applied this regression to plots in the following categories: (1) bench, (2) plateau, (3) low-severity, (4) mixed-severity, and (5) all *Ne Sextsine* to identify potential differences in changes to the fire history. Key dates of land use change were collated from a provincial study of fire governance (Copes-Gerbitz et al. 2022b), local histories and archival information (Day 1998, 2007, Mather 2000), and place-based oral histories of the *T'exelc* from *Ne Sextsine* (Copes-Gerbitz et al. 2021).

Results

Forest structure and composition

Psuedotsuga menzeisii accounted for 98% of trees sampled; a small component of *Picea engelmannii* x *glauca* (2%) and *Pinus contorta* (1%) was present in the plateau plots. Plots on the plateau were significantly ($P<0.05$) higher in median elevation (912 m.a.s.l) and on shallower slopes (5°) than plots on the bench (621 m.a.s.l and 14°, respectively; **Table 1**). Plateau plots were generally more south-facing whereas bench plots were more west-facing. Median canopy tree density was higher in plateau plots (159 versus 103 trees ha^{-1} in bench plots; $P=0.094$), while median subcanopy tree density was higher in bench plots (335 versus 287 trees ha^{-1} in plateau plots; $P=0.220$); however, these differences were not significantly different. Density of stumps was significantly higher ($P<0.001$) in plateau plots than bench plots (median 68 and 0, respectively). There was no significant difference in sapling or snag density between plot types ($P=0.532$ and $P=0.636$, respectively).

Fire occurrence, frequency and severity

At *Ne Sextsine*, samples from 118 fire-scarred trees yielded 189 indicators of fire (including 172 fire scars and 17 supporting scarlets); nine samples were excluded because of significant decay. Between 1553 and 1982 CE, 82 fire events were recorded (~19% of years; **Figure 2a, b, c**). Starting in 1628, the first year with ≥5 trees to potentially record fires, 35% of fire events burned at only one plot while 9% of fire events were widespread (1628, 1663, 1679, 1768, 1794, 1805, and 1870). No widespread fires were detected in three northeast fire plots (44, 58, 59).

Site-level and plot-level Weibull median fire intervals were 4 and 18 years, respectively; within plot intervals ranged from 2–189 years (Table 4.2). Two plots (44 and 59) had ≤2 intervals. Bench plots had a Weibull median fire interval of 22 years compared to 16 years for plateau plots, but the differences were not significant. Most fires occurred in the dormant season (late summer or autumn in these ecosystems; 56%) and the remainder occurred in the mid-summer (27% latewood) or early summer (11% earlywood); 17% of fire years (n=14) had scars in two different seasons (e.g., 1768 had two earlywood and four latewood scars that were considered two distinct fire events). Seasonality could not be determined for 6% of scars. Over the length of the fire record, the 1694 and 1727 earlywood fires occurred in one plot on the bench, while the others in the 1700s and 1800s occurred on the plateau. The earlywood fires in the 20th century occurred near an area of historical grassland in the southeast corner of *Ne Sextsine*.

Of the 1086 trees cored, we estimated the age of 1049 trees that established between 1585 and 1990; 37 trees were omitted because total age correction exceeded 25 years. The coring height correction ranged from 1–9 years (average 2±1 year), while the correction to pith ranged from 1–25 years (average 7±10 years). At the site-level density of tree establishment occurred in three modal age classes: 1810–1825 (926 trees ha^{-1}), 1840–1855 (2448 trees ha^{-1}), and 1885–1900 (4350 trees ha^{-1}; **Figure 2d**). The last mode accounted for 25% of the density of contemporary canopy trees.

At the plot-level, all 54 plots (both fire and age plots) had cohorts, and ten plots had two distinct cohorts (**Figure 3a, b and 4a, b**). Of the 64 total cohorts identified, 63% established asynchronously 10 to 50 years following the last widespread fire in 1870. In fire plots (**Figure 3a, b**), 20 cohorts were classified as fire-initiated and three of the five unattributed cohorts established after 1870. In age plots (**Figure 4a, b**), 25 cohorts were fire-initiated, 10 unattributed cohorts established after 1870, and four were undetermined because they were further than 2km from any sampled fire scars.

Ten fire plots were classified as low-severity (**Figure 3a**) and ten were classified as mixed-severity (**Figure 3b**) over the length of their plot-level fire records (**Table 2**); both classes included plateau and bench plots. No plots were classified as high severity because they all included fire scars. Each low-severity plot included 3–17 fires indicated by scars, plus ≥1 cohort(s) that persisted in the absence of fire. Seven of these plots had fire-initiated cohorts. The remaining three plots had one unattributed cohort with (n=1) or without (n=2) a second fire-initiated cohort. Two plots had unattributed post-1870 cohorts, and one plot had one fire-initiated and one unattributed post-1870 cohort. All low-severity plots had veteran trees ("scarred trees") that survived the cohort establishment. Each mixed-severity plot included 2–20 fires, plus at least one cohort that survived subsequent fire(s). Eight of these plots had fire-initiated cohorts. The other two plots had an unattributed cohort that established prior to 1870 with (n=1) and without (n=1) a second fire-initiated cohort. Plots categorized as low-severity had a Weibull median fire interval of 19 years compared to 18 years for mixed-severity plots, and these differences were not significant. In low- (mixed-) severity fire plots, seasonality of fire scars was 55% (64%) dormant, 15% (9%) earlywood and 30% (27%) latewood.

Of the age plots, 10 were classified as having low-severity (**Figure 4a**) and 20 were classified as having mixed-severity (**Figure 4b**) fire history through time. All low-severity plots were located on the western bench overlooking the Fraser River. Five low-severity plots had a fire-initiated cohort that persisted in the absence of fire, three of which also had an unattributed cohort that established after 1870. The remaining five low-severity plots each had only one unattributed cohort established after 1870. Of the 20 mixed-severity plots, all had a fire-initiated cohort. Two of these plots also had an unattributed cohort that established after 1870 but were considered mixed-severity because of the presence of the fire-initiated cohort that survived subsequent fires. Nine mixed-severity plots were on the bench and 11 were on the plateau.

Indigenous contribution to the fire regime

Piecewise linear regression detected four distinct periods of fire frequency that were generally consistent across and between plot type (bench vs plateau) and assigned fire severity through time (low- vs mixed-severity) (**Figure 5**). Across *Ne Sextsine* as a whole, fires were burning more frequently over the second period starting around the turn of the 19th century, indicated by the break points in 1787 (bench plots), 1804 (plateau plots), 1806 (mixed-severity plots) and 1825 (low severity plots). After this break point, fires become even more frequent during the third period. In the fourth period, in contrast, there is a clear decrease in fire frequency. The timing of this final break point is relatively consistent across all records, except in the low-severity plots when fire frequency begins to asymptote ~50 years earlier (1873; **Figure 5a**) than in the mixed-severity plots (1922). In contrast, the start of the asymptote is less than ten years apart between the plots on the bench (1905) and the plateau (1914; **Figure 5b**). At *Ne Sextsine*, the final break point (1916) is most similar in timing to that of the plots on the plateau (1914) and the mixed-severity plots (1922), suggesting that these plots are driving site-level cumulative fire frequency.

Discussion and conclusions

A mixed-severity fire regime

Historically, fires at *Ne Sextsine* burned frequently as part of a mixed-severity fire regime. This mixed-severity fire regime had a high proportion of low-severity fires indicated by fire scars that burned at local to widespread (for this study site) spatial scales, punctuated by patches of higher-severity fire indicated by cohorts at the local (plot)-scale only (Heyerdahl et al. 2012, Harvey et al. 2017). At the site-level, the mean fire return interval between 1628 and 1982 at *Ne Sextsine* (21 years) is consistent with mean intervals in other mixed-severity fire regimes in dry forest ecosystems in BC (Daniels and Watson 2003, Heyerdahl et al. 2012, Marcoux et al. 2013, Harvey et al. 2017, Brookes et al. 2021). Compared to mixed-severity fire regimes in higher-elevation montane forests in BC (Marcoux et al. 2013, Chavardès and Daniels 2016), the fire regime at *Ne Sextsine* contained little evidence of spatially extensive high-severity fire events. This contrasts the assumption currently guiding disturbance-based management in BC that spatially extensive high-severity fire events every c. 250 years were a component of mixed-severity fire regimes in dry forest ecosystems (BC Ministry of Forests and BC Ministry of Environment 1995) and helps to refine our understanding of the relative influence of low- to high-severity fire effects in dry forest ecosystems (Hessburg et al. 2016).

Traditionally in fire history research, even-aged cohorts are interpreted as indirect evidence of high-severity conditions that create large openings for groups of trees to establish, whereas a lack of cohorts and presence of fire scars is interpreted as evidence of low-severity fires (Heinselman 1973, Heyerdahl et al. 2012, Daniels et al. 2017). At *Ne Sextsine*, however, since all fire plots contained fire scars and cohorts, we interpreted cohorts that persisted in absence of fire to be a feature of low-severity fire histories at the plot level. This aligns with other interpretations of cohorts that persisted in the absence of frequent, low-severity fire as a legacy of high tree survival rates in altered fire regimes (Guiterman et al. 2018, Hagmann et al. 2021) rather than as evidence of a spatially extensive high severity fire. This criteria was also important at the nearby dry forest ecosystems in the (~100km south) Churn Creek Protected Area (Harvey et al. 2017) and the (~35km east) Alex Fraser Research Forest (Brookes et al. 2021). At *Ne Sextsine*, eight plots (**Figure 3a**) were classified as low-severity and contained a fire-initiated cohort that established after the last fire (Plots 15, 25, 45, 12, 14, 35) or was still forming after the last fire (Plots 28, 14). In these low-severity plots with 3–17 fire-scar years, we interpreted these cohorts as persisting in the absence of subsequent fire that would have likely killed small and vulnerable trees. Low-severity fires were also a feature of the mixed-severity fire plots, but the cohorts (both fire-initiated and unattributed) survived 1–11 subsequent fires that must have been of low enough intensity to not kill vulnerable young trees. This interpretation is supported by a clear reduction in fire-frequency, especially in the low-severity plots, starting in the 1870s and strong evidence of forest encroachment and increases in density in dry forest ecosystems in the absence of frequent fire in BC (Turner and Krannitz 2001, Harvey et al. 2017, Brookes et al. 2021) and similar frequent-fire ecosystems in North America (Hessburg et al. 2016, Larson et al. 2020, Hagmann et al. 2021).

In four fire plots (42, 44, 31 and 59) the cohorts were not attributed to a fire, but the timing of the cohort helped distinguish low-severity from mixed-severity fire history at the plot-level. We interpreted Plots 42 and 44 as low-severity because the cohort established after the last fire and persisted in the absence of fire, whereas we interpreted Plots 31 and 59 as mixed-severity because the cohorts occurred prior to fire scars and survived subsequent fires. We interpreted those unattributed cohorts in the mixed-severity plots (Plot 31 and the earliest cohort in Plot 59) as likely representing a high-severity fire event or patch of high-severity since these cohorts were preceded by no remnant trees. The two unattributed cohorts in the low-severity plots (42 and 44) were likely post-logging cohorts given the presence of stumps in those plots and their location close to historical bush mills where logging would have been prevalent (Day 1998, 2007, Mather 2000). It is possible that these unattributed cohorts were fire-initiated but no fire scar evidence remains because the contemporary or subsequent fires eliminated evidence, modern logging disturbance removed evidence, or due to the fading dendroecological record back in time (Swetnam et al. 1999). Alternatively, these cohorts could have resulted from rapid grassland invasion in the absence of fire given the historical presence of grassland at *Ne Sextsine* or were in response to favorable climate; however, we would expect synchronous establishment dates across plots if they were climate-driven rather than cohorts distributed through time (**Figure 2**). Nevertheless, these alternative explanations for the attribution of the cohort do not change our interpretation of low- versus mixed-severity fire histories at the plot-level or that the mixed-severity fire regime was dominated by low-severity fire.

In contrast to the fire plots, the age plots did not contain direct fire-scar evidence and, thus, were more challenging for interpreting fire severity through time. Our sample design distributed fire plots equally across the bench and plateau and all but four age plots (n=30) were within 2km of at least one fire plot from which we inferred severity. As with the fire plots, the timing of the cohort in the age plots was key for differentiating low- from mixed-severity. In the low-severity age plots (n=10), all cohorts (fire-initiated or unattributed) established at some point after the last widespread fire in 1870. However, in one of these plots (53) the cohort that initiated in 1905 did not persist in the absence of fire but was nevertheless interpreted as low-severity; this was because the fire that could have affected that cohort was only locally recorded at Plot 57 (versus at multiple plots) and because it occurred after the last widespread fire. All the mixed-severity age plots (n=20) had at least one fire-initiated cohort that may have survived (likely low-severity) subsequent fires. Through this approach, we may have overestimated the number of mixed-severity plots and underestimated low-severity plots because we relied on any adjacent evidence within a 2km radius (maximum of eight adjacent fire plots); if this is the case, low-severity fire was perhaps more frequent than our findings suggest. Some mixed-severity plots with only a single cohort and no veteran trees (e.g., Plots 13, 17, 20, 22, 37) may have experienced higher-severity fires or alternatively grassland invasion, especially since there were no fire scars visible at these plots that in the fire plots provide evidence of tree survivorship even during fire. Traditionally, the cohorts in these plots would have been interpreted as evidence of high-severity fire, however, given the amount of historical grassland present in this environment we think the latter explanation is more likely. It is notable that no low-severity age plots had this pattern. In addition, while this indirect evidence from adjacent plots and reliance on cohorts alone may not accurately represent the conditions within a plot given the variability of burn severities within a fire (Daniels et al. 2017), this approach, perhaps in combination with additional methods (e.g., spatial interpolation, Greene and Daniels 2017), may offer a possible alternative for fire-severity interpretations where it is not feasible to undertake intensive sampling of fire scars.

Indigenous contribution to the mixed-severity fire regime

Mixed-severity fire regimes are recognized for their complex interactions between topography, fuels and weather (bottom-up and top-down drivers) that help to create heterogeneity at different spatial and temporal scales (Schoennagel et al. 2004, Halofsky et al. 2011, Perry et al. 2011, Daniels et al. 2017). Inherent in contributing to this heterogeneity, although less well explored in studies on mixed-severity fire regimes, is the role of Indigenous land and fire stewardship. Oral histories and ethnographic research in BC describe the historical role of Indigenous burning and ongoing fire stewardship that has adapted to modern conditions (Turner et al. 2000, Lewis et al. 2018, Xwisten Nation et al. 2018, Lake and Christianson 2019, Dickson-Hoyle et al. 2021). Previous dendroecological studies have acknowledged the complex and context-specific relationships between Indigenous peoples and fire regimes in the USA (e.g., Stambaugh et al. 2013, Liebmann et al. 2016, Taylor et al. 2016, Guiterman et al. 2019, Larson et al. 2020, Kipfmueller et al. 2021) and Mexico (Fulé et al. 2011), but this is an emerging area of research within BC (Hoffman et al. 2017). Here, we discuss multiple lines of evidence used to interpret the role of Indigenous stewardship as a key driver of the mixed-severity fire regime at *Ne Sextsine.*

The complex distribution of fire severity at the plot-level is likely tied to diverse stewardship across *Ne Sextsine* since spatial variation in fire severity is not distinguished by contrasting topographic positions or forest (fuel) types. Oral histories, archaeological evidence, and ongoing collaborative research with the *T'exelc* indicate varied activities across *Ne Sextsine,* with areas of historically high occupancy centered at winter village sites and seasonal summer camps on the bench above the Fraser River (Copes-Gerbitz et al. 2021; Williams Lake Indian Band Traditional Land Use Team 1998). At and near these high use areas, constant campfires fueled by local wood sources were necessary for warmth, cooking, and drying fish in the late summer months. Berry-picking also occurred in open areas and forest edges near the Fraser River, where the berries would ripen earliest (Copes-Gerbitz et al. 2021); soapberry (*Shepherdia canadensis*) is a key medicinal and food species that can be maintained by frequent, low-severity fires (Walkup 1991). Hunting for mule deer was likely concentrated up on the plateau where there was more continuous forest cover (Copes-Gerbitz et al. 2021; Williams Lake Indian Band Traditional Land Use Team 1998); managing for mule deer winter range continues to be a guiding land-use objective (Steele et al. 2007). This complexity of land use likely increased the need for localized fires for different purposes, with 35% of fire years at *Ne Sextsine* recorded at only one plot. Similarly in fire regimes in the southwestern US, a high number of small fires is interpreted as an indication of Indigenous ignitions (Swetnam et al. 2016, Taylor et al. 2016).

These spatial patterns of use are also clearly differentiated in the plot-level fire severity: areas of high occupation on both the benches and the plateau are spatially coincident with plots classified as low-severity through time. The frequent use of intentional fire adjacent to village sites is a known practice in dry forest ecosystems and across the larger the *Secwépemc* traditional territory (Turner et al. 2000, Ignace et al. 2016, Lake and Christianson 2019). However, low-severity fire plots are also found in locations that were not previously identified as areas of high occupation, as recorded by the BC Archaeology Branch (W. Spearing, *personal communication*) or described by the *T'exelc* Elders (Copes-Gerbitz et al. 2021). The distribution of these low-severity fire plots beyond areas of known high occupation may be explained by several, not mutually exclusive, hypotheses: (1) not all archaeological sites have been recorded (Schaepe et al. 2020); (2) frequent, low-severity fire was used to maintain travel corridors (Lake and Christianson 2019, Larson et al. 2020, Kipfmueller et al. 2021) and to manipulate forest structure further from village sites (Lake and Christianson 2019, Roos et al. 2021); and/or, (3) low-severity fire could have resulted from lightning ignitions into areas of historical grasslands (Steele et al. 2007, Harvey et al. 2017). Although historical grasslands are present at *Ne Sextsine* (Steele et al. 2007), these are concurrent with known areas of high occupation. We therefore anticipate that hypotheses (1) and (2) are most likely, given the presence of culturally-modified way marker trees identifying travel corridors, the wide variety of stewardship by the *T'exelcemc* documented across *Ne Sextsine* both within and beyond areas of high occupation (Copes-Gerbitz et al. 2021), the visible evidence of previously unrecorded archaeological sites (Copes-Gerbitz et al. 2022c) and that all low-severity fire plots are within the boundaries of or within 300 meters of areas of high archaeological potential (Wilson et al. 1998).

Despite this likely association, the complexity of land use is mirrored in the complexity of fire-severity, unsurprising given that cultural burning has different objectives in different locations (Gottesfeld 1994, Lake and Christianson 2019) and Indigenous fire stewardship encompasses complex uses of and relationships to fire (Hoffman et al. 2022). For example, the low-severity plots were not just located along the bench of the Fraser River where most archaeological and oral history evidence is concentrated. Similarly, the mixed-severity plots were also within areas of high archaeological potential along the bench of the Fraser River. As these mixed-severity plots also have a high frequency of fire, we interpret the presence of mixed-severity plots as those where occupation may have shifted over time (e.g., intentionally leaving an area to recover temporarily stops frequent fire in the area), resource needs (and thus severity of fire) differed (e.g., forest-grassland edge requirements for berries), or areas that were purposely managed at lower intensities and may have been more amenable to lightning-ignited fires. Shifting objectives and fire use through time is a feature of Indigenous landscapes elsewhere in Canada (Davidson-Hunt and Berkes 2003), creating a dynamic mosaic of locally-intensive land use where fires may have been concentrated (Larson et al. 2020).

The seasonality of fire scars at *Ne Sextsine* also points to shifting objectives and land use through time and the complex pyrodiversity of Indigenous fire stewardship. Both spring and autumn burning during periods of low risk are common by Indigenous peoples in BC, depending on different objectives (Gottesfeld 1994, Lewis et al. 2018, Dickson-Hoyle et al. 2021, Hoffman et al. 2022). At *Ne Sextsine*, earlywood scars occurred in the areas of highest occupation on the benches in the early part of the fire record (late 1600s and early 1700s). We attributed these scars to spring burns, which could be evidence of cleaning village sites after overwintering (Huffman 2013) and before the rest of the seasonal round to other parts of their traditional territory (Copes-Gerbitz et al. 2021). Starting in the mid-1700s, earlywood scars were all situated on the plateau, and may be associated with renewing berry patches and ungulate forage, a common practice in the southern *Secwepemcúl'ecw* (Dickson-Hoyle et al. 2021) that is also consistent with the uses described by *T'exelc* Elders (Copes-Gerbitz et al. 2021).

Interestingly, for those scars in which we determined seasonality, the highest proportion (56%) were in the dormant season. Dormant season scars could indicate lightning-ignited fire during the period of highest fire risk in the region (Government of BC, 2021), the interpretation we used in our study. Alternatively, they could indicate intentional burning when fire risk was low (Gottesfeld 1994, Dickson-Hoyle et al. 2021, Hoffman et al. 2022). This latter explanation could mean that dormant fire scars reflect autumn burning of the year of ring formation or early spring burning prior to the onset of earlywood in the subsequent year. Dormant season scars have been attributed to early spring burning in other frequent-fire ecosystems with Indigenous fire stewardship (Guiterman et al. 2019, Kipfmueller et al. 2021), however, in those ecosystems modern fire frequency and lightning ignitions are also highest in spring. Uncertainty over the precise year and season of dormant fire scars makes it challenging to pinpoint whether individual fires were most likely ignited by Indigenous peoples or lightning. A priority for future research in our study area is to decipher fire seasonality and ignition sources to understand the full range of pyrodiversity; accurate allocation of fires to calendar years is key for analyzing fire-climate relationships, for example (Chavardès et al. 2022). Nevertheless, both intentional and lightning ignitions are a part of Indigenous fire stewardship (Miller et al. 2010, Hoffman et al. 2022) and we encourage future research to carefully consider how both types of ignitions may have been a part of an Indigenous-driven fire regime.

Perhaps the most compelling line of evidence indicating the historical fire regime resulted from Indigenous stewardship is the collapse of fire frequency during early colonization (**Figure 5**), a pattern that supports Indigenous oral histories in the region (Turner et al. 2000, Lewis et al. 2018, Xwisten Nation et al. 2018, Lake and Christianson 2019, Dickson-Hoyle et al. 2021). At *Ne Sextsine*, fire was excluded in the low-severity plots after 1873, corresponding to critical events associated with colonial impacts on Indigenous peoples in BC: the smallpox epidemic of 1862–63 which killed up to two-thirds of the *Secwépemc* peoples, regional pre-emptions that sold unceded land to settlers, and the federal Indian Act and Residential Schools program that forcibly removed Indigenous peoples from their territories and attempted to assimilate Indigenous children (Turner and Ignace 2000; Ignace and Ignace 2017, Chapter 12, pg.442). This was also the timing in which fire exclusion started to be legislated: in 1874 the BC Bush Fire Act in BC was passed and intentional fire use was punished, except by government authorization (Parminter 1981) which was unlikely to be granted to Indigenous peoples (Copes-Gerbitz et al. 2022b, Hoffman et al. 2022). These colonial impacts are also recent and ongoing, with the local residential school closing in 1981 and land sovereignty still retained by Indigenous peoples in the *Secwepemcúl'ecw* (Dickson-Hoyle et al. 2021), leading to continued tensions in land and fire governance (Verhaeghe et al. 2019, Dickson-Hoyle and John 2021, Copes-Gerbitz et al. 2022b, Hoffman et al. 2022). The collapse in frequent low-severity fire in areas of high Indigenous land use is a common pattern emerging in dry forests of BC (Brookes et al. 2021, Dickson-Hoyle et al. 2021) and in frequent fire regimes in other parts of North America where changes in social-ecological context are the primary driver of fire regime changes (Taylor et al. 2016, Larson et al. 2020, Hagmann et al. 2021).

In the mixed-severity plots, frequent, mostly low-severity fires continued until the 1920s. It was commonplace for early settlers to mimic Indigenous burning practices, although they often did so for different objectives such as clearing land for development and agriculture (Copes-Gerbitz et al. 2022b, Hoffman et al. 2022). Perhaps the continuation of frequent, low-severity fires in the mixed-severity plots into the late 1800s and early 1900s is a legacy of early settlers attempting to recreate more open forest conditions maintained by Indigenous stewardship in high-use areas. The frequent fires ceased around 1922, not long after policies of fire suppression began to take hold in BC (Copes-Gerbitz et al. 2022b) and Williams Lake was made divisional headquarters of the Pacific Great Eastern Railway, which brought an influx of European settlers, supported a boom in cattle ranching, and prompted the establishment of the first local bush mills in the 1940s (Day 1998, 2007, Mather 2000). Importantly, disentangling the drivers of altered fire regimes (e.g., exclusion of Indigenous fire stewardship vs. fire suppression) is key for identifying appropriate pathways to return pyrodiversity to these landscapes.

We acknowledge that Indigenous ignitions augmented a fire regime that also included lightning ignitions during years with high fire risk, such as those experienced in 2017. Historically, widespread fires in dry forests at the Churn Creek Protected Area (~37,000 ha; ~100 km southwest of *Ne Sextsine*) occurred during anomalously dry, warm years following antecedent drought conditions (Harvey et al. 2017) and regional synchrony in fire years corresponds with antecedent and current year drought conditions (Chavardès et al. 2022).

Notably, however, none of the four highly synchronous fire years in the dry forests of the Montane Cordillera Ecozone reported by Chavardès et al. (2022) were the same as the widespread fire years at *Ne Sextsine*. Only one highly synchronous fire year (1831) was also classified as a widespread fire year at the nearby Alex Fraser Research Forest (~30km southeast of *Ne Sextsine*; widespread defined as ≥25% of plots including fire scars; Brookes et al. 2021) and Churn Creek Protected Area (Harvey et al. 2017); in contrast, only 10% (n=2) of plots at *Ne Sextsine* included fire scars that year. Locally, of the 23 fire years recorded at the nearby Alex Fraser Research Forest (Brookes et al. 2021), 12 also occurred at *Ne Sextsine.* Common fire years could mean that regional drought was favourable for synchronous fires (Chavardès et al. 2022) or that similar patterns of intentional fire use were occurring in both places, given that the Alex Fraser Research Forest is also within the *T'exelc* traditional territory. Drought-induced fire events tend toward higher-severity effects (Agee 1993, Schoennagel et al. 2004), but even the most widespread fires at *Ne Sextsine* experienced only low- to moderate-severity effects at the plot scale. For example, the 1794 fire which scarred 56% of recording plots, resulted in a cohort in only one plot (38).

The prevalence of historical low- to moderate-severity fires provide evidence of the self-limiting nature of frequent fire regimes in Indigenous territories (Stambaugh et al. 2013, Swetnam et al. 2016, Taylor et al. 2016) and the importance of Indigenous stewardship in reducing the likelihood of negative impacts from future fires that could burn at uncharacteristic high severity (Christianson 2015, Trauernicht et al. 2015, Bird et al. 2016, Mistry et al. 2016, Taylor et al. 2016, Hoffman et al. 2021, Mariani et al. 2021). This is true not only of intentional ignitions, but also other stewardship practices such as wood collection (Roos et al. 2021), which was an important practice supporting salmon drying and staying warm through the winter at *Ne*

Sextsine (Copes-Gerbitz et al. 2021). Importantly, interpreting the fire regime of *Ne Sextsine* without differentiating low- from mixed-severity plots through time, or only focusing on common bottom-up (topographical differences between the plateau and bench) or top-down (climate) drivers, may have led to the interpretation that the Indigenous fire regime was not disrupted by colonization and instead continued until more intensive grazing and active fire suppression occurred. Differentiating between low- and mixed-severity fire at the plot scale provided a key piece of evidence that supports Indigenous oral histories that describe diverse stewardship and disruption of that stewardship due to colonization (Lake and Christianson 2019).

Culturally and ecologically relevant landscape management

At *Ne Sextsine*, we explicitly recognize that Indigenous stewardship contributed to the patch heterogeneity that characterizes mixed-severity fire regimes (Perry et al. 2011). After colonization and the collapse of the fire regime at *Ne Sextsine*, the landscape transitioned into a dense, homogenous forest mostly devoid of Indigenous stewardship, with nearly two-thirds of cohorts establishing in the century after 1870. This interpretation of forest dynamics contrasts the common mis-interpretation that high-severity fires produced cohorts at the end of the 1800s (Brookes et al. 2021) and that fire suppression altered fire regimes in other mixed-severity fire regimes (Chavardès and Daniels 2016). The homogenization of landscapes under altered fire regimes is a common pattern in mixed-severity fire regimes (Perry et al. 2011, Hessburg et al. 2016) and fire regimes across North America (Hagmann et al. 2021) that can result in increased vulnerability to uncharacteristic, large, high-severity fires. This vulnerability is enhanced under current and future climate change that facilitates a warming and drying trend (and subsequent lengthening of the fire season) in the southern Canadian cordillera (Wang et al. 2017, 2020, Hanes et al. 2019), with dry forest ecosystems especially at risk when modern forest structures do not provide resistance or resilience to crown fires (Turner 2010, Johnstone et al. 2016, Hessburg et al. 2019).

Given the degree of change from historical forest conditions and ongoing climate change impacts, many advocate for learning to coexist with fire through adaptive management approaches that incorporate diverse strategies to deal with risk (Day and Pérez 2013, Christianson 2015, Hessburg et al. 2016, Smith et al. 2016, Coogan et al. 2019, Prichard et al. 2021). These diverse strategies must reflect the unique forest context, including a refined understanding of the location-specific mixed-severity fire regime and the relative influence of low- to high-severity fire events through time (Stephens et al. 2013, Hessburg et al. 2016). In BC, this context includes reflecting on the accuracy of the Natural Disturbance Type system that underrepresents the prevalence of mixed-severity fire regimes and overestimates the contribution of high-severity fire (Marcoux et al. 2013). Although we did not aim to specifically reconstruct event-level fire severity, the results from *Ne Sextsine* demonstrate that at least 98% of fires were of low or moderate-severity and that high-severity fires were not a feature of the forest historically.

In addition, addressing fire risk must be weighed against other forest values, such as where managing for mule deer winter range may influence forest susceptibility to insects and fire (Leclerc et al. 2021). Ecologically-based forest management in mixed-severity fire regimes suggests creating and maintaining successional heterogeneity at multiple scales and expanding the use of prescribed and managed wildfire (Hessburg et al. 2016). Beyond these ecologically-informed strategies, however, future management must first recognize the contribution of Indigenous peoples to historical fire regimes and include ecocultural strategies such as cultural burning and a return of Indigenous land stewardship if we are to minimize the high-severity impacts such as those experienced directly adjacent to *Ne Sextsine* in 2017 (White et al. 2011, Christianson 2015, Lake and Christianson 2019, Dickson-Hoyle et al. 2021).

Central to future management of these ecocultural landscapes is embracing the diversity of relationships between people and fire, and explicitly recognizing how this relationship has changed through time (Bowman et al. 2011, 2013, Moritz et al. 2014, Smith et al. 2016, Hessburg et al. 2019). Indigenous fire management has historically contributed to pyrodiversity (Huffman 2013, Trauernicht et al. 2015) and despite the ongoing impacts of colonialism, continues to do so (Eriksen and Hankins 2014, Martínez-Torres et al. 2016, Lake and Christianson 2019), including areas where fire was intentionally used or where Indigenous peoples benefitted from fire (Miller et al. 2010, White et al. 2011, Hoffman et al. 2022). The interpretations made from dendroecological and other palaeoecological methods can potentially erase or misconstrue the relationship between Indigenous peoples and fire when not undertaken in collaboration with Indigenous scholars, Elders and other knowledge keepers – which became clear in a recent debate in the literature around the interpretation of Indigenous fire use in the northeastern USA (Abrams and Nowacki 2020, Leonard et al. 2020, Oswald et al. 2020, Roos 2020). To understand historical fire regimes, it is imperative to overcome the duality of fire as 'natural' or 'cultural' (Bowman et al. 2011, White et al. 2011, Roos 2020) and, instead, embed a process that meaningfully collaborates with Indigenous peoples and supports Indigenous-led management approaches that reflect their understanding of the dynamic role of fire and people in the landscape (Lake 2013, Eriksen and Hankins 2014, Mistry and Berardi 2016, Lake and Christianson 2019, Larson et al. 2020, Dickson-Hoyle et al. 2021). These placed-based approaches are key for enabling the adaptive capacity of communities (Roos et al. 2016) and for providing pathways towards Indigenous revitalization (Kimmerer and Lake 2001, Lake and Christianson 2019, Dickson-Hoyle et al. 2021). In this way, future fire regime research can more holistically incorporate diverse perspectives necessary for coexisting with fire in a culturally and ecologically relevant way (McWethy et al. 2019, Larson et al. 2020) and ensure future natural science research is led by Indigenous peoples (Mistry and Berardi 2016, Wong et al. 2020).

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Tables

Table 1: Plot and tree characteristics from plateau and bench forest plots at *Ne Sextsine*.

Median (range) presented for plot characteristics and stand structure. Slope aspect transformed to 0–180° (NE–SW; cool-warm) scale. P-values are the result of Mann-Whitney U tests of median values, except for slope aspect which is a result of a two-tailed t-test.

	Median (range)		
Plots	**Plateau (n=20)**	**Bench (n=34)**	**p–value**
Plot Characteristics			
Elevation (m.a.s.l)	912 (693–967)	621 (477–859)	P<0.001
Slope Aspect	100° (3°–175°)	122° (8°–173°)	P=0.288
Slope Angle	5° (1°–35°)	14° (1°–34°)	P=0.019
Stand Structure			
Canopy trees/ha	159 (39–496)	103 (30–397)	P=0.094
Sub-canopy trees/ha	287 (70–882)	335 (58–1705)	P=0.220
Saplings/ha	300 (0–1000)	200 (0–1800)	P=0.859
Snags/ha	7 (0–196)	11 (0–328)	P=0.636
Stumps/ha	68 (0–355)	0 (0–129)	P<0.001

Table 2: Characteristics of the fire record for 20 fire plots in *Ne Sextsine*. Plots are stratified by the final fire severity through time and arranged from the shortest to the longest fire record within severity type. Order corresponds to plots in Figure 2 A (low-severity) and B (mixed-severity). Persistence of cohort "yes" indicates that one or more (either both or just the second/later) cohorts persisted in the absence of fire at the plot-level.

			Fire return intervals			Fire severity evidence			Fire severity through time
Plot Number	Location	Fire record	No. intervals	Weibull median (yr)	Range (yr)	No. fire-initiated cohorts	No. unattributed cohorts	Persistence of cohort	
14	Plateau	1848–1982	4	34	22–43	1	0	Yes	Low
12	Bench	1757–1841	6	14	8–22	2	0	Yes (both)	Low
42	Plateau	1719–1902	14	12	6–32	0	1	Yes	Low
25	Bench	1694–1870	9	19	8–33	1	0	Yes	Low
28	Plateau	1628–1904	16	16	2–38	1	0	Yes	Low
33	Plateau	1603–1870	13	20	10–40	1	0	Yes	Low
45	Plateau	1679–1870	7	18	5–115	1	1	Yes (both)	Low
44	Plateau	1553–1919	2	--	--	0	1	Yes	Low
35	Bench	1673–1901	6	28	2–98	1	0	Yes	Low
15	Bench	1606–1840	6	28	8–134	1	0	Yes	Low
59	Bench	1848–1919	1	--	--	1	1	Yes (second)	Mixed
18	Plateau	1818–1915	9	10	5–22	1	0	No	Mixed
16	Bench	1768–1901	6	20	9–54	2	0	Yes (second)	Mixed
38	Plateau	1794–1909	4	26	10–51	1	0	No	Mixed
58	Bench	1684–1893	9	21	6–51	1	0	No	Mixed
32	Plateau	1663–1902	15	14	2–40	1	0	No	Mixed
57	Bench	1688–1935	11	21	13–59	1	0	No	Mixed
30	Bench	1614–1943	6	35	2–189	2	0	No	Mixed
31	Bench	1693–1904	6	20	2–110	0	1	No	Mixed
27	Plateau	1554–1898	19	16	3–51	1	0	No	Mixed

Figure captions

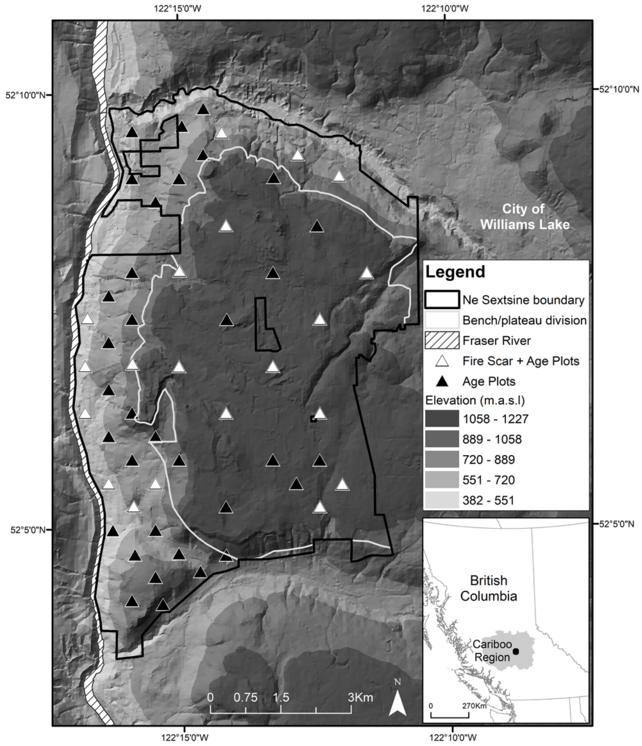
Figure 1: Location of 20 fire and age plots (white triangles) and 34 age-only plots (black triangles) at *Ne Sextsine*, Williams Lake Community Forest. Inset: Location of *Ne Sextsine* (black dot) in British Columbia, Canada.

Figure 2: Fire regime of *Ne Sextsine*. Horizontal lines represent plots by fire history through time as low-severity (A) or mixed-severity (B). The study site composite (C) summarizes the percentage of plots that burned relative to the number of fire plots that could have recorded fire ("recorder depth") in each fire year. Forest age structure (D) indicates the density of tree establishment in each 15-year age class.

Figure 3: Fire history and forest demography in (A) low-severity fire plots (n=10) and (B) mixed-severity fire plots (n=10). Each plot includes plot-level fire events (triangles), density of tree establishment (grey bars), presence of cohorts (vertical lines), and sample depth of scarred trees (hashed). Plot numbers in green/orange are located on the plateau/bench.

Figure 4: Forest demography in (A) low-severity age plots (n=10) and undetermined age plots (n=4) and (B) mixed-severity age plots (n=20). Each plot includes density of tree establishment (grey bars) and presence of cohorts (lines). Plot numbers in green/orange are located on the plateau/bench.

Figure 5: Temporal shifts in cumulative fire frequency at *Ne Sextsine* relative to key colonial events for (A) low-severity, mixed-severity, and all fire plots (*Ne Sextsine*), and (B) bench, plateau, and all fire plots (*Ne Sextsine*).



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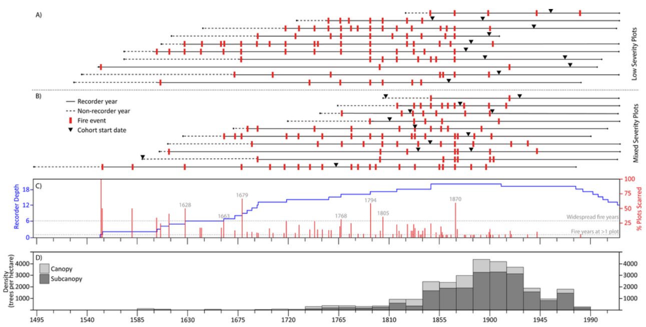

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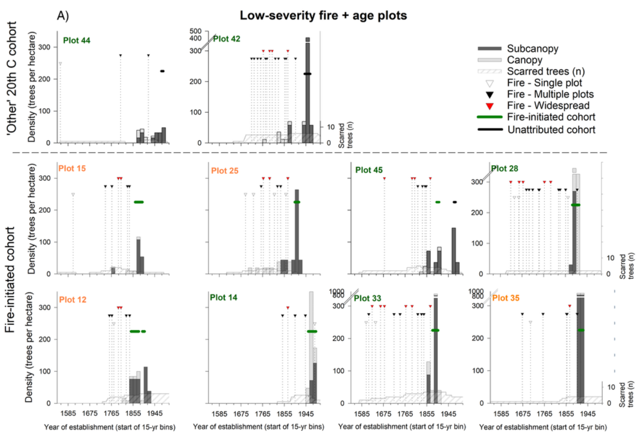

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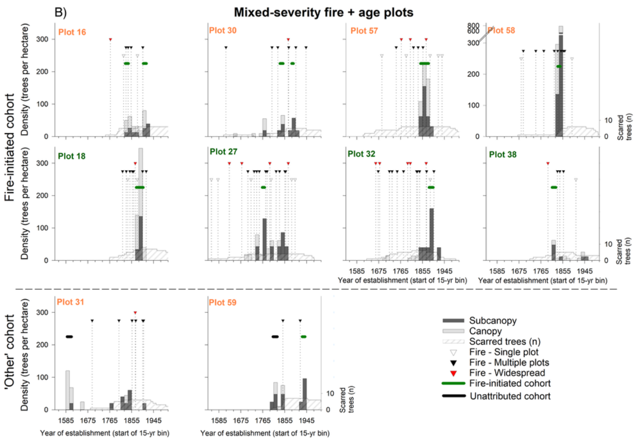

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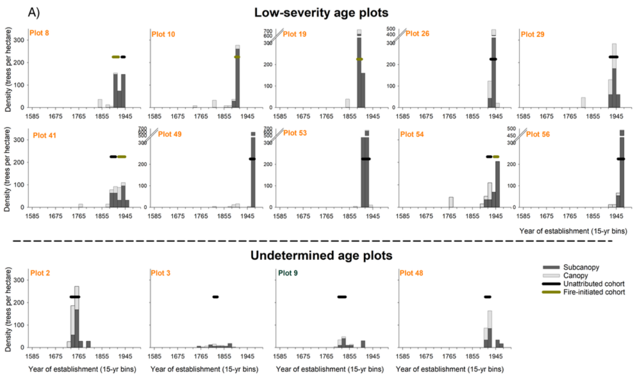

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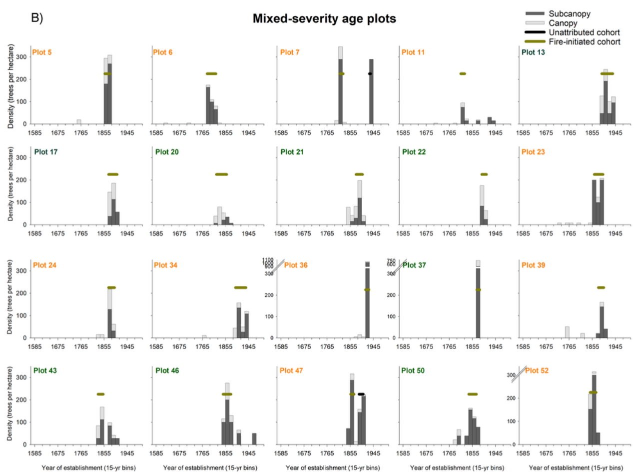

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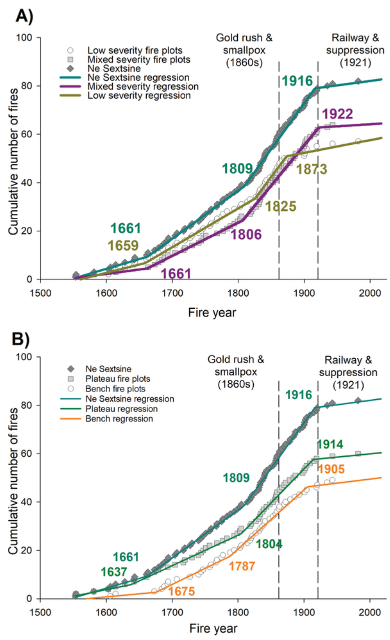

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