Forest Ecology and Management 432 (2019) 840–849

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

A nation-wide analysis of tree mortality under climate change: Forest loss and its causes in Israel 1948–2017

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ARTICLE INFO

Keywords:
Drought
Forest fire
Insect attack
Snow damage
Forest dieback
Hydraulic vulnerability

ABSTRACT

Is tree mortality increasing? Are recent mortality events related to climate change? Which tree species are the most affected? Many case studies have been published in the last decade, but the necessary large-scale and long-term knowledge is still missing.

Here we combined data from forest surveys and satellite imagery, to create the first spatial tree mortality history at the national scale. Israel is a small country with only 7% forest cover, but its large environmental diversity and mosaic of many, small, forest areas makes it a good 'miniature model' for the task.

Tree mortality events have been increasing significantly since 1991 and correlated well with drought. Among mortality events, 24% of the loss was directly related to drought, and 58% to fire, with 69% of fires occurring over a drought background. Conifers were disproportionally more affected than native broadleaved trees.

This is the first national-scale study of tree mortality dynamics, and it confirms the suspected increase in this phenomenon in recent decades, and the dominant role of drought. Our study opens a way to a better, multi-source monitoring future for forest management and ensuring forest sustainability under climate change.

1. Introduction

The future of global forests under climate change is a pertinent, fundamental question in ecology, terrestrial biogeochemistry, and beyond. Governments and land managers work together with forest ecologists and tree physiologists to understand the underlying mechanisms of tree mortality, produce reliable predictions, and take action to ensure the sustainability of forests in decades to come (Zeppel et al., 2011; Hartmann et al., 2015, 2018). In some areas, the mission seems more urgent than in others (Klein et al., 2014b; van Mantgem et al., 2009). For example, parts of South West North America have been suffering large-scale drought-induced tree mortality in the past decade (McDowell et al., 2008; Adams et al., 2009). In South West Australia, extreme drought and heat caused forest canopy collapse (Matusick et al., 2013), highlighting the vulnerability of those forests to tipping points (Laurance et al., 2011). At the same time, large forest areas have been lost to the mountain pine beetle epidemic in North West North America (Bentz et al., 2010), and to wildfires in Russia (de Groot et al., 2013), both related to climate change trends of increased winter and summer temperatures, respectively (Klein, 2015). In contrast, other areas, including Europe, Eastern North America, and tropical Africa, report little mortality (but see Neumann et al., 2017). Yet climate change-induced growth and productivity enhancements intensify competition at the stand scale, in turn leading to higher mortality (Stephenson et al., 2011; Luo and Chen, 2015).

Following the reviews of existing tree mortality reports (van Mantgem and Stephenson, 2007; Allen et al., 2010, Greenwood et al., 2017; Young et al., 2017), we generally agree that drought-induced tree mortality is globally increasing (Allen et al., 2015; McDowell and Allen, 2015, but see Steinkamp and Hickler, 2015). However, this statement assumes three pieces of information: (1) the knowledge of tree mortality history over decades; (2) the measurement of the area of forest loss, i.e. the quantification of direct observations; and (3) the evidence that mortality was the result of drought, rather than other potential factors. In essence, none of the three assumptions has been sufficiently tested beyond the stand scale. While important datasets have been compiled, they are based on anecdotal reports, which are mostly subjective (Allen et al., 2010 and references therein), rather than on long-term, wide-scale surveys.

The problem of causality is even more complex (Sala et al., 2010;

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https://doi.org/10.1016/j.foreco.2018.10.020
Received 30 August 2018; Received in revised form 5 October 2018; Accepted 8 October 2018
0378-1127/

McDowell, 2011; Klein et al., 2014b; Anderegg et al., 2016). A prerequisite to the claim that drought-induced tree mortality is increasing is evidence that tree mortality follows drought episodes, and reduced at times without drought (Luo and Chen 2013, 2015). This is, however, hard to obtain, because: (1) tree response times are species- and ecosystem-specific (e.g. Mueller et al., 2005; Klein et al., 2014a); (2) lagged and delayed responses exist and are further complexed by inter-annual variability, forming long-term 'drought legacies' (Schiller, 1977; Anderegg et al., 2013; Cailleret et al., 2017); and (3) the involvement of other interacting factors, such as fire and insect attack creates complex syndromes, rather than coherent, cause-effect links (de Groot et al., 2013; van Mantgem et al., 2013). Attempts to discriminate between causes have been focused on climate interactions and on deciphering the physiological mechanisms (Klein et al., 2011; McDowell, 2011; Adams et al., 2017). Anderegg et al. (2016) have used divergence in species responses to investigate the cause of death at the plant trait level, showing fingerprints of hydraulic effects, in turn highlighting the role of drought. Such trait-based approaches can help informing the selection of tree species for future afforestations, as well as identifying forests at higher risk. For example, McDowell and Allen (2015) predicted that tall, old-growth conifers are at greater risk of loss than broadleaf species.

At present, the lack of temporal and spatial data on forest tree mortality is inhibiting an unbiased view and an informed discussion of future options. Sources for such data are not within the scientific milieu, but rather in the hands of forest management agencies, governments, and agencies operating remote sensing tools. Data flow between these entities is essential for constructing a tree mortality database (Hartmann et al., 2018). Ideally, such a database should be at the global scale, feeding on nation-scale data. For example, the condition of forests across Europe has been monitored continuously and coordinated through the ICP Forests network (Lorenz, 1995), established in response to reports of air pollution-induced tree mortality. Still, the European monitoring, which is arguably more advanced than that of similar networks outside Europe, relies on a finite number of plots, rather than covering entire forest areas (Michel and Seidling, 2016). If a monitoring plot is situated outside a tree mortality event in a forest, no record is being kept. As in many research areas, the application of remote sensing tools holds the potential to revolutionize the field. Satellite imagery has been used already in multiple studies of tree mortality (e.g. Dorman et al., 2015a, 2015b; Stevens-Rumann et al., 2018) and should increase in the future, as analyses continue to develop (e.g. Hansen et al., 2013 and the Global Forest Watch; www.globalforestwatch.org/map). However, the heterogeneous distribution of tree mortality on the one hand, and the limits in satellite spatial resolution and coverage on the other hand, mean that surveys on ground will still be needed (Hartmann et al., 2018).

Here we combined historical series of forest inventory data, aerial photography, and satellite imagery, to create the first dynamic tree mortality map at the national scale. Our research objective was to test the assumption of increasing drought-induced tree mortality, and to provide a nation-wide, chronological dataset with insights on the different causes of forest loss, and the interspecific differences in vulnerability. Specifically, the following hypotheses were tested: (1) Over the past decades, tree mortality has increased; (2) The increase in tree mortality is related to drought, either directly or via interaction with fire and insect outbreak; and (3) Among species, pine, cypress, and eucalypt species are more vulnerable to drought than other, broadleaved species.

2. Materials and methods

2.1. Study area

Israel is a small country with only 7% forest cover, but its large environmental diversity and mosaic of many small forest areas makes it a good 'miniature model' for the task. Moreover, the national scope goes beyond the spatial size, since forests worldwide are managed at a national scale. In addition, the region around Israel (Middle East; Europe) has so far been underrepresented in the global analysis of tree mortality (Allen et al., 2010). Most forests are concentrated around three major areas: the Galilee and Carmel in the north, west of Jerusalem in the center, and north of Beer Sheva in the south (Fig. S1). With the withdrawal of the Ottoman Empire in 1918 and the establishment of Israel in 1948, deforestation stopped, and afforestation has been intensified (Osem et al., 2008; Osem, 2013). Today, Israel has ~80,000 ha of planted forest: 50,000 ha of conifers, mainly the native *Pinus halepensis* and, to lesser extent, other Mediterranean pine and cypress species; and 30,000 ha of broadleaved, e.g. eucalypt (mostly *Eucalyptus camaldulensis* and *E. gamphocephala*), olive and carob. In addition, there is ~80,000 ha of low-stature maquis (mostly oak and pistacia), which has been recovering from centuries of deforestation and over-grazing. The planted forest area increased from 27,900 ha in 1960 to 50,100, 63,700, and 75,600 in 1970, 1980, and 1990, respectively. With time, early afforestations matured, while younger afforestations were added. Causes for tree mortality in Israel include fire, drought, insect outbreak, and also snow storm damage to non-adapted tree species (Fig. S2). Wind storms and pathogens are nearly nonexistent and have negligible effect.

2.2. Data sources

To produce the most complete dataset of forest tree mortality possible, multiple data sources were used, combining data from ground surveys and from remote sensing analyses (Table 1). These sources are divided into eight categories: (1) Published forestry data, as reported in the bulletins of Israel's forest agency, KKL-JNF. These bulletins have been published (in Hebrew) since 1953, and are now available online (http://www.kkl.org.il/afforestation-and-environment/publications/). The primary forestry bulletins were 'LaYaaran' (1953–1985), 'LaYaaranim' (1991–1995), and 'Yaar' (2002-ongoing; Yaar = Hebrew for 'forest'). (2) Secondary forestry reports, including the technical journal 'Yaaron' (1976–1978), and scientific papers published in *Ecology & Environment* (2010-ongoing). Overall, bulletin reports were the most important source in our analysis, offering a wealth of information (Table 1; see '*Data harmonization and standardization*' below). (3) Special, internal reports of the KKL-JNF, issued following the 1999 and

Table 1
Data sources used in the analysis of tree mortality in Israel 1948–2017. KKL-JNF is the Forest Service of Israel. The total number of mortality events is smaller than the sum of reports due to multiple data sources for a single event in some cases (e.g. remote sensing and ground survey).

Data source		Timeframe	Number of reported mortality events
KKL-JNF primary forestry bulletins	'LaYaaran'	1953–1985	48
	'LaYaaranim'	1991–1995	0
	'Yaar'	2002–2017	32
KKL-JNF secondary forestry reports	'Yaaron'	1976–1978	4
	Ecology & Environment Magazine	2010–2017	9
KKL-JNF special forestry reports		1999, 2009	19
KKL-JNF website		2015–2016	6
KKL-JNF maps website		2016–2017	8
Israel's natural disaster research database		1978–2016	19
Dorman et al. (2015a,b)		2011–2012	4
Global Forest Watch Satellite data		2000–2016	14

2009 drought events. (4) Data published on the KKL-JNF website, e.g. reporting the outcome of the forest fires in 2015 and 2016. (5) Data published on the KKL-JNF maps website (http://kkl.maps.arcgis.com/home/index.html), e.g. the analysis of forest fire effects in 2016 and 2017. (6) Data on forest fires published in Israel's natural disaster research database of the University of Haifa (https://geo.hevra.haifa.ac.il/~inh/index.php/en/). (7) Published ground and satellite-driven data in Dorman et al. (2015a, 2015b). (8) Satellite data downloaded from Global Forest Watch (GFW; www.globalforestwatch.org/map) for forest loss during 2000–2016. Data for the region studied is based on Hansen et al. (2013). We used the map tool with 10% minimum canopy density; tree cover for 2000; and tree cover loss for individual years between 2001 and 2016. The size of forest loss events was determined by GFW analysis tool. Forest loss events larger than 10 ha were included in the dataset. All data sources were screened for quality and consistency, and incomplete data were discarded from the analysis.

Forestry data were collected by KKL-JNF foresters on ground (no aerial survey) as part of routine and *ad-hoc* surveys. In such surveys, all trees with crown height > 1 m height are included. The cause of tree mortality was assigned by the Chief Forester at the region level (North, Center, and South). Snow- and fire-induced mortalities were typically obvious, whereas drought- and insect-induced mortality cases were sometimes harder to differentiate. In case of ambiguity, expert opinion was given by a forest entomologist of the forest service (KKL-JNF) or belonging to the Ministry of Agriculture, Natural Resources unit. Ultimately, we report the direct cause of mortality, rather than its predisposing factors. To investigate the role of such factors, we use the dataset to test the contribution of climate in fire- and insect-driven mortalities.

2.3. Testing for sampling bias in the dataset

A dataset can provide a good representation of event magnitudes at their spatial and temporal contexts, as long as sampling is performed in a coherent, constant manner (Lewis et al. 2004). In our case, these conditions relate to reporting, rather than sampling. To this end, we illustrated the number of forestry reports published per year in 1953–2017 (Fig. S3). We included primary (1), secondary (2) and special (3) reports, as defined above. Forest monitoring has been performed at the national scale consistently, and hence the spatial coverage has been constant across all study years. Notably, gaps exist during the late 1980′s and the late 1990′s. However, there was no indication that these gaps caused any reporting-induced bias; on the contrary, there were more reports of tree mortality events which occurred during these periods than, for example, during the early 1980′s or early 2000′s. Plotting annual forest area loss against the number of reports per year showed no relationship between the two (Fig. S3). Specifically for drought-induced mortality events, records from drought years in the 1960′s and early 1990′s (Table S1) negate the potential claim that the larger events observed since 1999 reflect a sampling bias rather than a real trend.

2.4. Data harmonization and standardization

At source, reports of tree mortality events were made in one of three ways: (1) forest area lost (ha); (2) the percent loss of forest area in a specific forest site; and (3) the number of dead trees. To permit a quantitative and comparative study, all data were transformed into forest area, in ha. To clarify, a 10 ha forest area loss could be the result of a tree mortality event at that exact magnitude, or, for example, a 100 ha forest with 10% mortality (10% × 100 ha = 10 ha), i.e., an area-equivalent mortality rate. In case of percent of forest area, the transformation was allowed using case-specific gross forest area from GFW using the online analysis tool. In some cases, insect-induced mortality was based on a sampled area within a forest, and the transformation from percent to ha was performed by upscaling the results using tree cover area from GFW. When the number of dead trees was reported, transformation to area loss used assumptions on stand density. According to KKL-JNF data the mean stand density of 600, 500, and 300 trees ha^{-1} was assumed for north, central, and south forests, respectively. These stand densities represented mostly mature forest stands, which have already reached final height and leaf area index. Finally, to neutralize the effect of changes in forest cover during the study period, all data are also reported as forest loss (% area) in the supporting information.

Mortality patterns are often spatially heterogeneous. For example, drought-induced tree mortality can have a diffuse pattern, making remote sensing detection very difficult (Hartmann et al., 2018). In the case of diffuse mortality pattern, we multiplied the affected forest area by the observed percent of dead trees. In the case of snow damage, we distinguished between tree breakage and tree collapse, accounting only for the latter as indicative of tree mortality. In the case of fire-induced mortality, large corrections were applied. Often, the gross area affected by the fire was registered, including non-forested areas and areas with limited fire damage, e.g. forest floor fire, with little effect on tree crowns. To normalize for these effects we took advantage of GFW observations, representing the conservative estimate of forest area loss. Specifically, three forest fire events have been used for calibration purposes. In the 2010 Carmel fire, the gross area affected and reported by the KKL-JNF was 2500 ha, whereas the satellite-detected net forest loss was 1016 ha. In the 2005 fire in Aminadav, the gross area affected was 120 ha, of which 22 ha were detected by the satellite observations. The 2013 fire in Yehiam was followed by clearance and replanting of 225 ha, of which 60 ha were lost on the GFW satellite-based map. Based on these ratios, a calibration factor of 0.29, reflecting an average ratio, was used for normalizing fire reports to tree mortality areas.

Tree species identification was reported for some mortality events, e.g. for species-specific insect attacks. In addition, partitioning into species were done in surveys of the 1999 drought-induced mortality and in analysis of the 2017 fire-induced mortality. In cases where species identification was missing, this information was obtained from the national masterplan for forests and forestry.

2.5. Climate data

Total precipitation and mean temperature at the diurnal temporal scale were obtained for three meteorological stations representing the three major forested areas in the North (Zefat), Center (Jerusalem), and South (Beer Sheva) of the country, for the period 1950–2017. These data were downloaded from the Israel Meteorological Service Database (https://ims.data.gov.il), along with information on snow depth in single snowstorm events in Zefat and Jerusalem. Precipitation anomalies were calculated for total annual precipitation, as the difference between a value in a given year and the long-term average (1950–2017). Temperature anomalies were calculated for the annual average diurnal Tmean, as the difference between a value in a given year and the long-term average (1971–2000).

To integrate precipitation and temperature effects into a single climate index, a Standardized Precipitation-Evapotranspiration Index (SPEI) was calculated at the monthly temporal scale. Using SPEI, drought periods were delineated, also showing drought severity, and accounting for the combined effect of warming and drying (Adams et al., 2009; Allen et al., 2010, 2015; Anderegg et al., 2013). Using monthly observations of precipitation (P) and potential evapotranspiration (PET), SPEI calculates standardized deviations from the average monthly local water balance (P - PET). PET is estimated based on minimum and maximum temperatures and latitude. The SPEI equations have been previously reported (Vicente-Serrano et al., 2010). Here we used a 12-month integration period, which showed higher sensitivity of local pine forests than other tested periods (Dorman et al., 2015a). Data were downloaded from the global SPEI database (http://sac.csic.es/spei/) for three 0.5^{0} grid cells representing Israel's major

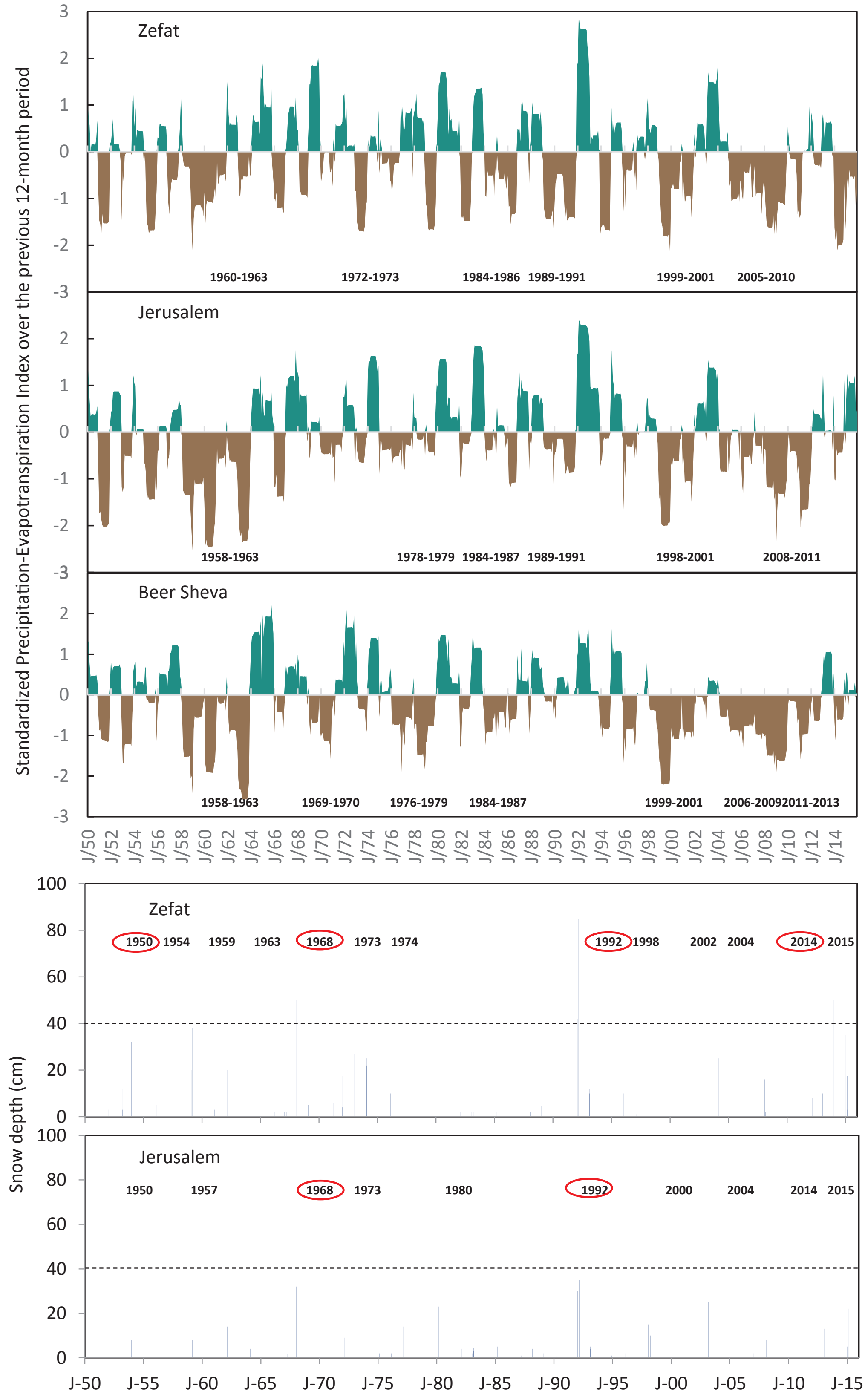

Fig. 1. Drought index changes (top) and snowfall events (bottom) over Israel's forest areas in 1950–2017 (J = January). Positive and negative values of Standardized Precipitation-Evapotranspiration Index over the previous 12-month period (SPEI12) denote water surplus and deficit, respectively. Zefat, Jerusalem, and Beer Sheva data represent north, central, and south forests, respectively (snowfall in Beer Sheva was negligible). Years with consecutive drought (top) or large snow storms (bottom) are indicated.

forest areas: Zefat (32°45′N, 35°15′E), Jerusalem (31°45′N, 35°15′E), and Beer Sheva (31°15′N, 34°45′E). This spatial resolution was sufficient, since each of the three cells matched its respective forest area, and within 25, 3, and 5 km from its meteorological station at Zefat, Jerusalem, and Beer Sheva, respectively.

2.6. Testing the predictability of tree mortality in forests in Israel

To test whether the insights gained in our analysis can be applied to predict tree mortality events in the future, a simplified predictive model was constructed. The model was set to predict the magnitude of tree mortality events, calculated as forest area loss, as a function of climatic factors (drought index at current, 1 year, and 2 years prior to mortality, annual precipitation, annual anomaly of the average diurnal Tmean, or snow depth in case of snow-induced mortality), and the development of afforestation areas. The six climatic variables were selected based on the observation that short- and long-term trends in Tmean were similar with those of Tmin and Tmax (Israel Meteorological Service); data not shown); the drought index integrated P and PET observations; and the negligible role of wind in tree mortality. Our simplistic approach did not explicitly account for tree-tree competition, tree size and age, despite of their demonstrated role in tree mortality (Lutz and Halpern, 2006, Young et al., 2017). First approximation formulations were tested on cause-specific mortalities. These mechanistic formulations followed the following form:

$$A = \alpha C D^{\beta} \tag{1}$$

where A is the annual forest area loss (ha year^{-1}), α and β are empiric factors calculated from an optimization step, C is a climatic factor, and D is a forest area development factor, calculated as the fraction of total afforestation area from the maximum value (in the year 2000). Interdecadal changes in D were assumed as linear. $\beta \geq 1$ was registered to account for positive feedbacks generally associated with the increase in planted forest area, such as increases in tree age and canopy cover as forests mature. Discrete models were constructed for each mortality cause, with α and β values optimized to obtain the highest Pearson correlation coefficient for a 1:1 regression line between observed and predicted forest area loss values.

2.7. Statistical analysis

Unlike many past analyses of tree mortality, as well as models and estimates of other forest disturbances, our all-inclusive analysis excludes estimates or sampling. Instead, we bring definite values of tree mortality as recorded by foresters and forest scientists for the entire temporal and spatial frames (1948–2017 in Israel). For this reason, the values reported are free of any uncertainty term, e.g. standard deviation.

Analysis of variance (ANOVA) for forest and climate effects on the extent of tree morality was performed in cases where the number of cause-specific events (n) permitted. Considering $n = 4$, 4, 20, and 29 for snow-, drought-, insect-, and fire-induced tree mortality, only the latter two were included in the analysis. ANOVA was performed in JMP (Cary, NC, USA) at $\alpha = 0.05$.

3. Results

3.1. Climate trends over Israel's forests 1948–2017

Situated in the Mediterranean climate zone, Israel's forests are affected by both temperature and rainfall patterns. Mean annual precipitation amounts of the past decades were significantly lower during several periods: late 1950′s- early 1960′s, mid 1980′s, late 1990′s, and late 2000′s (Fig. S4). Other periods of consecutive below-average rainfall years occurred in some areas and not in others. For example, the late 1970′s were exceptionally dry in the south, and recent years in the north were mostly dry (Fig. S4). Following a similar trend, the annual average of mean diurnal temperature indicated a ~1 °C warming in the late 1950′s- early 1960′s, and the late 1990′s, increasing to ~1.5 °C warming in the recent decade, across the three forest areas. Taken together, drying and warming were usually coupled, as were, conversely, wetting and cooling. To integrate rainfall and temperature effects, we calculated the standardized precipitation-evapotranspiration index at the monthly scale for each of the three areas (Fig. 1). The period since 1999 had the highest rate of water deficit months, creating an almost continuous drought period in Beer Sheva and Jerusalem, and the most severe drought on record in Zefat. Some north and central forests are at elevations of > 700 m, and hence might receive snowfall every other year. The snow storms of 1950, 1968, 1992, and 2014 were exceptionally large, with snow depths of 40 cm or more. Snow storms typically occurred during years of positive water balance, and hence drought and snow risks were usually alternating and not occurring at the same year.

3.2. Temporal changes in tree mortality and tree mortality causes

Overall, across 158 records of tree mortality, 6531 ha of forest were lost over seven decades (Table S1). The rate of tree mortality varied considerably from one decade to the other: 42% and 31% of the loss were in the 1990s and 2010s, respectively, whereas other decades had only 3–8% of the loss. Notably, both these decades were drier than most of the other decades, i.e. had relatively hotter and drier years, with negative water balance (Fig. 2). The period of 1958–1963 was exceptionally dry, yet at that time the total forest area was smaller, and the trees were younger (Fig. S5). Only fire-induced mortality has been reported for that period. In addition, the more recent droughts (1999 and later) were accompanied by a strong, ongoing, warming trend (Fig. S4). Calculating the forest loss as percent of the planted forest area, pre-1980 mortalities are highlighted (Fig. S5). Still, only the 1999 drought and the 2010 fire caused losses > 1%.

The five largest mortality events were the 1999 drought-induced mortality across Israel (1268 ha), the 2010 Carmel fire (1016 ha; the largest single-site forest loss), the 1992 snow damage in montane forests (408 ha), the 1995 Shaar hagay fire (364 ha), and the 1991 pine bast scale mortality in north and central Aleppo pine forests (244 ha; the largest single-species mortality event). In total, causes for tree mortality partitioned between fire (3800 ha), drought (1550 ha), snow (839 ha), and insect outbreaks (356 ha).

3.3. Spatial patterns of tree mortality

Expectedly, the majority of mortality events were concentrated around the three major forest areas in the north, center, and south of the country's Mediterranean zone (Fig. 3). Yet, the distribution of mortality events has been growing, especially since the 1990′s. Until the 1980′s, most of the mortality events were concentrated around the Carmel and Jerusalem forests. In the period of 1985–1999, tree mortality expanded into forest sites in the upper Galilee (South of the Lebanon border), the North-East, and the South-West. Notably, most of this expansion was driven by drought-induced mortality. In recent years, forests South-West of Jerusalem and in western Galilee were also affected, mostly by fire-induced mortality (Fig. 3).

3.4. Interaction between drought and other tree mortality causes and the predictability of tree mortality in forests in Israel

Is drought involved as an underlying driver of other types of tree mortality? A correlation between SPEI12 and the size of fire- and insect-induced tree mortality events indicated that 54% of insect attacks and 61% of forest fires occurred during years of water deficit (data not shown). Considering lagged drought effects, 76% of bast scale mortalities correlated with water deficit in the preceding year, and 69% of

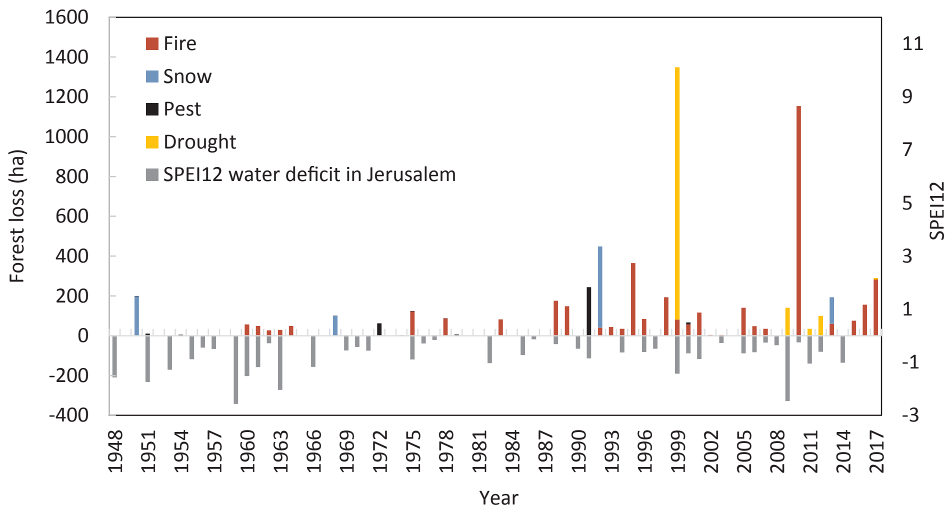

Fig. 2. Tree mortality in Israel, in ha of forest area lost, over the past seven decades, partitioned by year and cause. Water deficit years are indicated by the Standardized Precipitation-Evapotranspiration Index over the previous 12-month period (SPEI12 < 0, in Jerusalem; up to 2015).

fires correlated with water deficit two years before the fire (Fig. 4). These ratios indicated the possibility of drought acting as a predisposing factor to the immediate killer, be it fire or insect attack. Although our dataset spans over seven decades, the small number of cause-specific events for some mortality causes prevented further analysis. Fire and pine bast scale were the only mortality causes providing sufficient data for a meaningful analysis of variance, considering six climate and forest parameters (Table 2). The extent of fire-induced mortality was significantly affected by temperature, and moreover, by the water deficit one year prior to mortality. None of the effects on pine bast scale was significant.

To what extent can we predict the magnitude of forest loss in a given year? We tested whether the insights gained in our analysis can be applied in a simplified predictive model, based on climate and afforestation area development, as summarized in Eq. (1), $A = \alpha CD^{\beta}$. Discrete models were constructed for each mortality cause, using snow depth and the five climatic parameters in Table 2 to parameterize C. Among the four mortality causes, snow-induced mortalities, and, to lesser extent, drought-induced mortalities, showed a relatively good predictability ($r^2 = 0.22–0.23$; Fig. S6). In the two models, A is the forest area loss (ha year^{-1}), C is the sum of snow depths in Jerusalem and Zefat, when one of them > 40 cm (snow) or SPEI12 (drought), D is a forest area development factor, calculated as the fraction of total afforestation area from the maximum value, and α and β are 3.5 and 1, respectively (snow) or 88 and 3 (drought). Clearly, these indications are preliminary, as they rely on a very low number of events. The equation we developed for snow-induced forest area loss predicted the 1992 and 1968 snow-induced mortalities relatively well (5% and 55% errors, respectively; Fig. S6), while overestimating the 2013 damage and underestimating the 1950 damage (225% and 45% of observed losses, respectively). In predicting drought-induced tree mortality, the 1999 major drought event was excluded from the analysis, since its impact was disproportionally larger than could have been predicted by any model formulation. The 1999 drought mortality was unexpected, especially considering the harsher water deficit of 2009, with far smaller consequences (Fig. 2). Using our equation, the latter forest loss was overestimated by 23% only (Fig. S6). The 2011 mortality was also overestimated, whereas the 2012 mortality was underestimated (207% and 41% of observed losses, respectively). Our preliminary model predicted many relatively small drought impacts over the years (up to 75 ha), which were absent in our records. The 1999 mortality event was captured in the model as the first major drought impact, however at a fraction of its observed magnitude (122 vs. 1268 ha; not shown). Multiple model formulations constructed for the prediction of fire- and insect-induced mortalities failed in finding predictable patterns, in line with the high distributions of these events in relation to climate (Fig. 4).

3.5. Differential vulnerability among tree species

Partitioning the mortality events among tree species showed a clear pattern of higher vulnerability in the planted forest, especially of pine and eucalypt species, in comparison with high resilience of the maquis oaks and other broadleaved tree species (Fig. 5). This partitioning is remarkable, especially considering the larger area occupied by native broadleaves than by conifers. *Pinus halepensis*, and the related *Pinus brutia* were involved in more than half of the mortality events. Insect outbreaks have been mostly reported in the early 1950′s, through the 1970′s, and again in 1991 and 2000 (Fig. S7). Aside from anecdotal reports, almost all insect attacks were on pine species, with larger effects of the pine bast scale (*Matsucoccus josephi*), and smaller effects of pine bark beetles, such as *Orthotomicus erosus* and *Pityogenes calcaratus*. Fatal insect attacks were also reported for *Pinus pinea*, as well as for *Quercus calliprinos* and *Arbutus andrachne* in 1963, in combination with drought (Table S1). Pines and eucalypts were also the major victims of the 1999 drought-induced mortality (Fig. S8), with similar species partitioning in southern and northern forests. In addition, 47% and 43% of *Pinus halepensis* and *Eucalyptus* mortality events, respectively, were single-species mortalities, compared to 24% and 5% of events involving *Quercus calliprinos* and *Cupressus sempervirens*, respectively. These rates suggest that monoculture plantations were more vulnerable than mixed forests. The sclerophyllous broadleaved species of the Mediterranean maquis proved to be very drought-resistant, as could have been expected (Schiller et al., 2003). Most of these species are capable of resprouting, and hence mortality can be apparent, rather than final (Sever and Neeman, 2008). Native broadleaves (including fruit trees and maquis species) were almost entirely missing from the 1999 drought-induced tree mortality event, but did account for 23% of the fire-induced mortalities for a given year (2017; Fig. S8). Again, considering the larger area covered by broadleaves, their smaller part in forest losses highlights their higher resistance, even to fire. The higher fire risk of conifers and eucalypt might be related to higher stature, drier wood, and higher flammability (e.g. terpene-containing resin).

4. Discussion

Our work provided the first nation-scale analysis of tree mortality history. Within the study's spatial and temporal context, large-scale drought-induced forest dieback was a relatively new phenomena, which

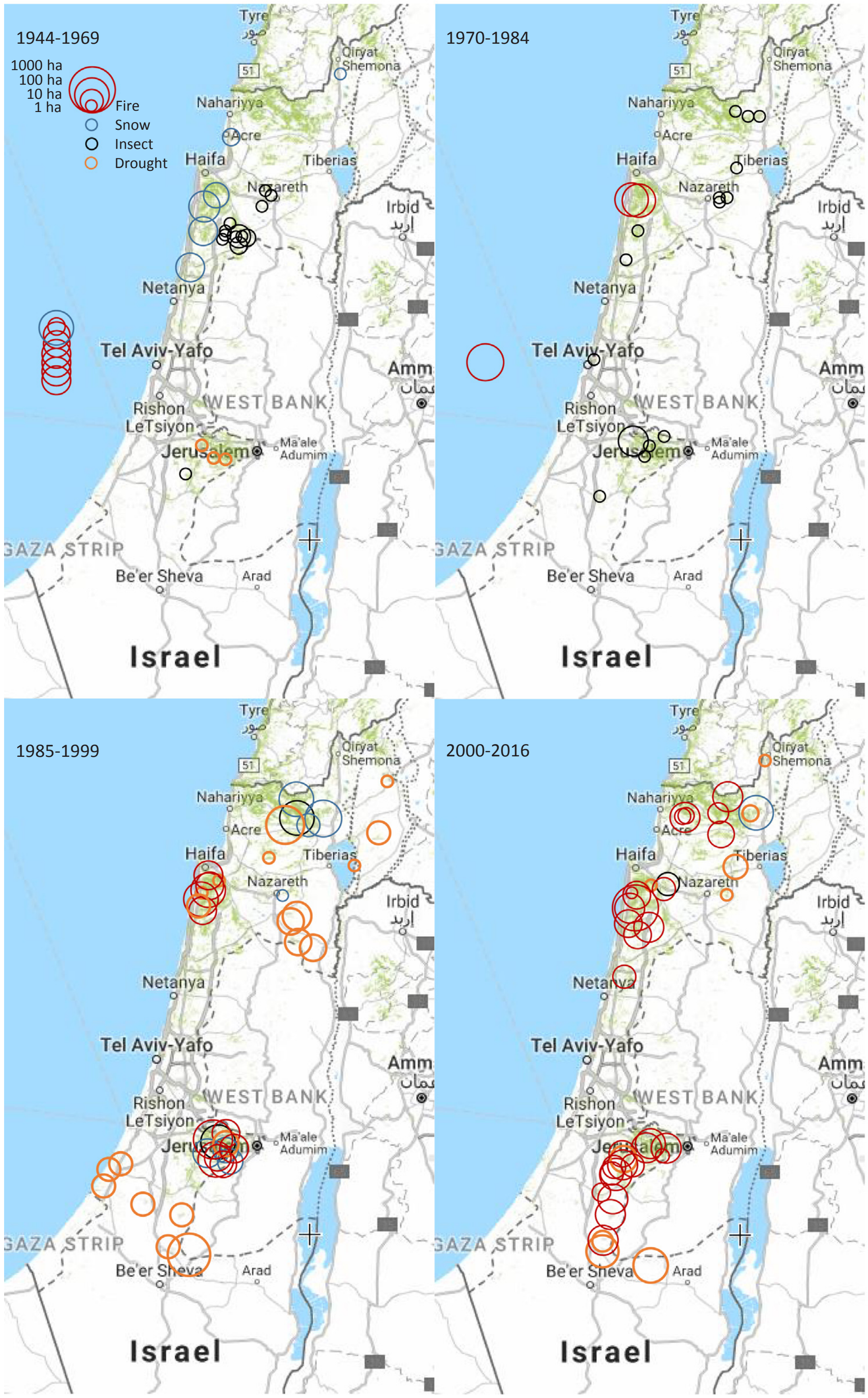

Fig. 3. Spatial distribution of tree mortality events in Israel for 15 years' time windows (except for 1944–1969; 26 years). Forest areas appear in green. Circle size and color denote event size (forest area lost) and cause, respectively. Circles marked over the Mediterranean Sea denote nation-scale data without spatial partitioning.

appeared in 1999, and then again, to smaller extent, in 2009, 2011, and 2012 (Fig. 2). Drought-induced tree diebacks have been already reported in 1963, 1991, and 1994, though at local scale (Table S1). The year of 1999 had the lowest annual precipitation on record (Fig. S4), and the year of 2009 had the lowest SPEI12 (Fig. 2), combining the warming and drying effects. Therefore, our data confirms the assumption of increasing drought-induced tree mortality, and more mortality can be expected in future drought years (McDowell and Allen, 2015; Greenwood et al., 2017). These results are in line with previously published measurements and estimates for North America (Adams et al., 2009; Allen et al., 2015), also highlighting the role of warming in more recent droughts, i.e. that contemporary, hotter droughts are more fatal than droughts in previous decades. Due to Israel's high diversity of ecosystems, our conclusions have the potential for extrapolation into

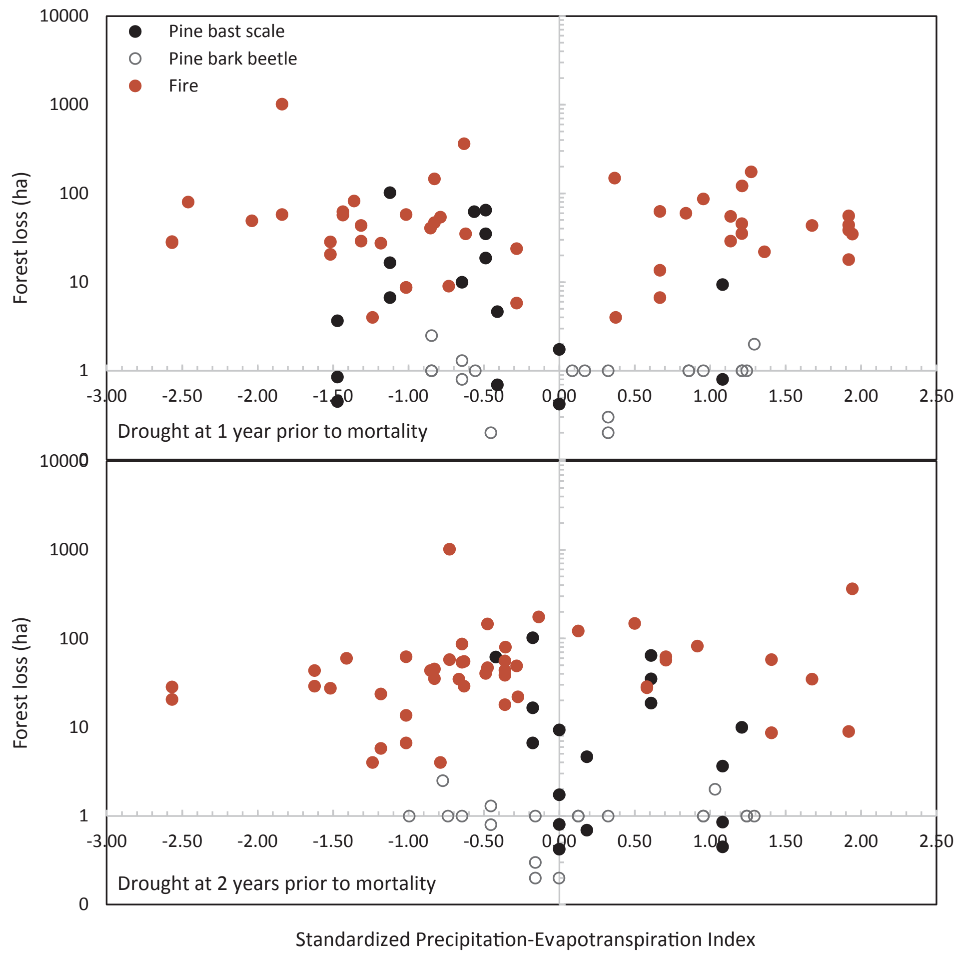

Fig. 4. Correlation between fire- and insect-induced tree mortality events and Drought Index changes. Positive and negative values of Standardized Precipitation-Evapotranspiration Index over the 12-month period (SPEI12) one year (top) or two years (bottom) prior to mortality, denote water surplus and deficit, respectively, at the region of tree mortality site (North, Center, or South). Note the logarithmic scale on y-axis.

Table 2
ANOVA results for forest and climate effects on fire-induced and pine bast scale-induced tree mortality in Israel 1948–2017. Annual SPEI (Standardized Precipitation-Evapotranspiration Index) values include water deficit years only. Significant effects at α of 0.05 are in **boldface**.

Factor	Fire-induced mortality		Pine bast scale-induced mortality	
	t Ratio	Prob > t	t Ratio	Prob > t
Annual precipitation	0.37	0.7126	−1.18	0.2423
Annual mean temperature anomaly	2.37	**0.0213**	−1.35	0.1822
SPEI	0.28	0.7805	−0.26	0.7946
SPEI-1	−2.96	**0.0044**	−0.72	0.4720
SPEI-2	−0.05	0.9563	0.69	0.4899
Forest development index	1.64	0.1069	1.09	0.2804

multiple biomes, including semi-arid, dry Mediterranean, and wet Mediterranean. Indeed, similar trends have been recently reported for Mediterranean Europe (Neumann et al., 2017), as well as important Mediterranean climate regions overseas, e.g. in South West Australia and in California (Matusick et al., 2013; Baguskas et al., 2014, respectively). The combination of ground surveys and remote sensing data proved synergistic: while identification of pre-2000 and diffuse mortality patterns required the ground surveys to be recorded, satellite data provided the best record for large-scale forest area loss annually. By comparing between the two approaches, we were able to normalize the gross fire area into net fire-induced forest area lost.

Regarding the first hypothesis, we confirm that over the past decades, tree mortality has been increasing. The 1991 pine bast scale epidemic marked a new era in Israel's forests, with annual losses surpassing the 200 ha (Fig. 2). One reservation to this conclusion is that, as in many other forests globally, Israel's forests are part of a highly dynamic landscape, driven by land-use change, overriding natural changes. It is still possible that the combination of forest area increase and forest maturation, set a lag time by which mortalities in the past three decades were related with the maturation of trees, which were mostly planted in 1960–1990. Yet, the increase in forest area has been moderated since 1990, and hence it cannot be stated that the increase in forest area loss was directly driven by the increase in total forest area (see also Table 2). Concerning the age effect, pine bast scale mortality events in the 1970′s are a good example, since the insect attack was mostly on 20–30 years-old trees (Mendel et al., 1997; Osem, 2013). But age had a lesser effect than drought, as evidenced by the gap in mortality records in 2002–2004, which had mostly water surplus (Fig. 2). It should be also noted that the oldest planted forests in Israel are < 100 years, and less than a third of all planted trees are older than 60. Our second hypothesis is partially supported: the increased mortality was related to drought, but overall, fire-induced mortality was the major death cause. Yet, the majority of fire events were related to drought, typically with a 2-year lag time (Fig. 4), as observed in SW USA (van Mantgem et al., 2013). The 2010 Carmel fire, which followed two consecutive drought years and started after eight consecutive

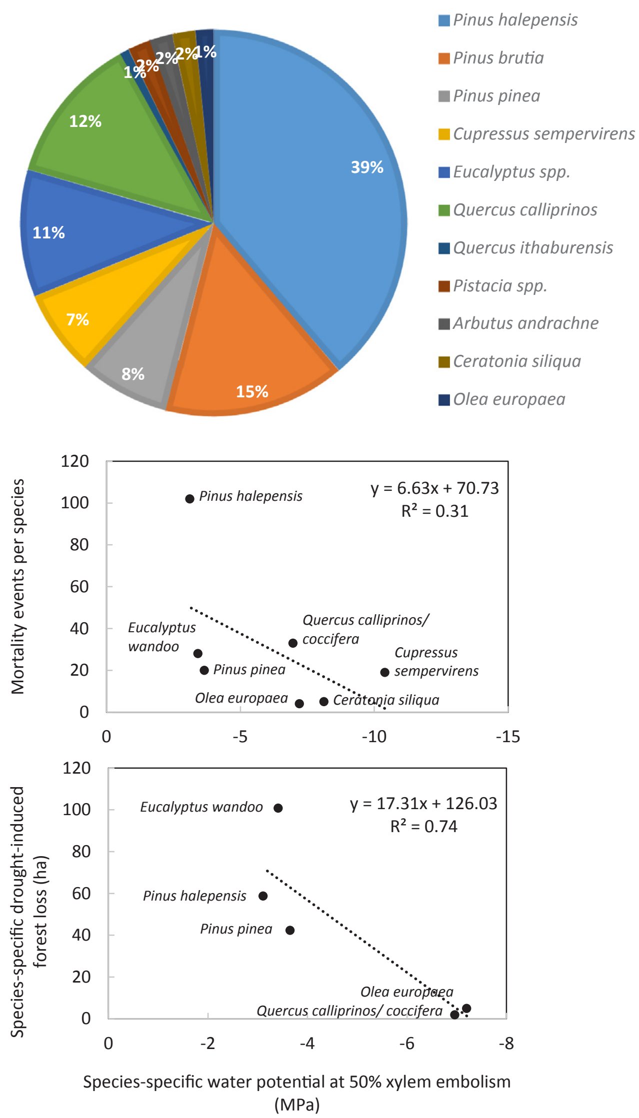

Fig. 5. Top: Tree species partitioning in tree mortality events in Israel along the past seven decades. Partitioning corresponds to the number of events where the species has been affected. The scatterplots show the relationship between tree species occurrence in mortality events (middle: by number of events; bottom: by species-specific forest loss in the 1999 drought) and its xylem hydraulic vulnerability, as measured by water potential at 50% loss of hydraulic conductivity due to cavitation. Water potential data from Choat et al. (2012).

months of zero precipitation, accounted for > 15% of the entire forest loss in 1948–2017. The link between pine bast-scale mortality and climatic stress at the previous year probably reflect a known infection lag (Dietze and Matthes, 2014). In spite of these interactions, there was little overlap among mortality types in a given year (Fig. 2). Fire- and insect-induced mortalities followed additional drivers other than drought. For example, some of the bark beetle infestations were due to thinning activities.

Fire-induced forest loss can be hard to predict (van Mantgem et al., 2013; Stevens-Rumann et al., 2018). This can be due to complex fire dynamics, involving stochastic wind dynamics. Even if fire intensity and extent could have been predicted, the impact on trees involves additional degrees of freedom such as tree status, flame height, and more. Last, the level and response time of firefighting intervention can make the difference between minor and major forest loss, as was the case of the 2010 Carmel fire. Similarly, pest management activities (Fig. S2), when efficient, can stop an insect outbreak, thereby inhibiting prediction of insect-induced forest loss. In addition, complex insect population dynamics further complicate any prediction (Dietze and Matthes, 2014). Our observation of the effects of temperature increase and drying on fire-induced mortality (Table 2) is in line with declining forest resilience to wildfires under climate change (Stevens-Rumann et al., 2018). The large 2010 fire coincided with the exceptional warming conditions in that year (Fig. S4). Notably, drought- and snow-induced tree mortalities are definite when occurring, and cannot be controlled in the short-term. Still, the (partial) success of our simplistic model in predicting the occurrence and magnitude of these mortality events is worth noting. These predictions overcame multiple sources of variance, including stand density and elevation, as well as tree species and age. The relatively accurate prediction of snow-induced mortality indicates the quasi-linear effect of snow depth in this mortality-type mechanism, with 40 cm snow as a mortality threshold (Fig. S6). The magnitude of drought-induced mortality was also predictable, excluding the major 1999 event. Here, the merit of our preliminary modelling exercise is in highlighting the non-linear nature of tree mortality events. Drought-induced mortality might act at both proportionate and tipping-point manners (Steinkamp and Hickler, 2015). For example, the 2009 drought impact could have been predicted using an equation predicting other drought impacts (2011, 2012), while the 1999 drought impact seems like a tipping-point event.

Last, the large species divergence (Fig. 5) confirms our third hypothesis of the increased vulnerability of conifers and eucalypt compared to native broadleaves. The divergence between the planted forest trees and the maquis trees can be partly explained by forest management issues, such as planting of inadequate conifer provenances, excessively high stand density, or delayed thinning (Schiller, 2013). Also, monoculture plantations showed higher mortality ratios than mixed forests, stressing the role of biodiversity in ecosystem resilience, with implications to forest management. Nevertheless, the involvement of multiple tree species in drought- and fire-induced mortalities (Fig. S6) indicates a strong drought signal, rather than a scenario of planted forest collapse. Anderegg et al. (2016) compared coexisting tree species in mortality events on a global scale, and showed that higher mortality can be expected based on xylem vulnerability to cavitation. Here, we used published species-specific data on hydraulic vulnerability (Choat et al., 2012), assuming little environmental plasticity, to test for a potential link with mortality. Species-specific mortality was expressed as either the number of mortality events per species, or the forest loss area in the 1999 drought event. The pine and eucalypt species which had higher mortality rates, had indeed higher hydraulic vulnerability than that of the cypress and the native broadleaved (Fig. 5). This good correlation between mortality and xylem cavitation supports the conclusion of Anderegg et al. (2016), while highlighting the involvement of drought sensitivity in mortality *per se*, including fire- and insect-induced mortality. In a comparative study of *Pinus halepensis* and *Quercus calliprinos*, the local forest and maquis dominant tree species, respectively, the tighter stomatal regulation of the pine was related to its lower water-use compared with the oak (Klein et al., 2013). Here we show that hydraulic constraints are better predictors of a tree species' mortality outcomes along decades and across regions.

5. Statement of authorship

Data collection was performed by all authors and coordinated by RC and TK. TK performed the data analysis and wrote the manuscript.

6. Data accessibility statement

All data reported are included in the Tables, Figures, and Supporting Information.

Acknowledgements

The authors wish to thank the Merle S. Cahn Foundation and the

Monroe and Marjorie Burk Fund for Alternative Energy Studies; Mr. and Mrs. Norman Reiser, together with the Weizmann Center for New Scientists; and the Edith & Nathan Goldberg Career Development Chair. The authors thank two anonymous reviewers for their useful comments which helped improving the paper.

Competing interests

The authors declare no competing financial interests.

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2018.10.020.

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