Environ Monit Assess (2023) 195:1478
https://doi.org/10.1007/s10661-023-12060-5

RESEARCH

Tracking 20 years of forest demographics in east Texas, USA, using national forest inventory data

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Received: 14 August 2023 / Accepted: 27 October 2023 / Published online: 15 November 2023

Abstract Forest resource reporting techniques primarily use the two most recent measurements for understanding forest change. Multiple remeasurements now exist within the US national forest inventory (NFI), providing an opportunity to examine long-term forest demographics. We leverage two decades of remeasurements to quantify live-dead wood demographics which can better inform estimates of resource changes in forest ecosystems. Our overall objective is to identify opportunities and gaps in tracking 20 years of forest demographics within the US NFI using east Texas as a pilot study region given its diversity of tree species, prevalence of managed conditions, frequency of disturbances, and relatively rapid change driven by a warm, humid climate. We examine growth and mortality rates, identify transitions to downed dead wood/litter and removal via harvest, and describe implications of these processes focusing on key species groups (i.e., loblolly pine, post oak, and water oak) and size classes (i.e., saplings, small and large trees). Growth and mortality rates fluctuated differently over time by species and stem sizes in response to large-scale disturbances, namely the 2011 drought in Texas. Tree-fall rates were highest in saplings and snag-fall rates trended higher in smaller trees. For removal rates, different stem sizes generally followed similar patterns within each species group. Forest demographics from the field-based US NFI are informative for identifying diffuse lagged mortality, species- and size-specific effects, and management effects. Moreover, researchers continually seek to employ ancillary data and develop new statistical methods to enhance understanding of forest resource changes from field-based inventories.

Keywords Forest resources · Forest Inventory and Analysis (FIA) · Long-term data · Snag-fall · Tree-fall · Tree demography

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10661-023-12060-5.

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Introduction

The frequency and intensity of natural disturbances have increased relative to historic levels in forest ecosystems, changing growth and mortality rates, decomposition processes, and the accumulation, transfer, and emissions of tree-based carbon in forests (Sommerfeld et al., 2018). Tree demographic rates are at the heart of biomass and carbon fluxes within forested ecosystems, as it is the change in status of individual stems that ultimately informs the movement of these materials. Carbon estimation and greenhouse gas emissions estimation (e.g., the US Greenhouse Gas Inventory, EPA, 2022) rely, in part, on data from the US national forest inventory (NFI) which is administered and executed by the US Department of Agriculture Forest Service, Forest Inventory and Analysis (FIA) program (McRoberts et al., 2005). For over two decades (since the 1998 Farm Bill), the FIA program has applied a largely standardized inventory protocol across all states and territories of the USA. Agencies such as the National Aeronautics and Space Administration (NASA) and the US Geological Survey (USGS) have invested in monitoring programs (e.g., orbiting carbon observatories, space-based lidar) that provide research platforms for refining our understanding of global carbon dynamics. Yet, FIA is uniquely suited to inform forest resource management, including carbon, given the detailed plot, site condition, and tree-level information across tree sizes (e.g., seedling to mature trees) collected and rigorous tracking and remeasuring of individual plots and trees over the last two decades (Tinkham et al., 2018). However, there remains a need to understand how FIA's quantification of long-term demographics can provide greater context and resolution to temporal changes in forest resources.

Estimates of live and dead wood volume, biomass, and carbon stocks in US forest ecosystems are currently obtained from models relying on tree measurements and site-level characteristics available on all core FIA plots (i.e., live and standing dead trees) or on a subset of FIA plots (i.e., downed dead wood) (Domke et al., 2022). Collectively, these estimates represent a field-based measurement of the live and coarse dead wood organic matter at each point in time. Changes in these forest ecosystem pools are then inferred over measurement cycles via remeasurements at discrete points in time. This approach, broadly defined as the "stock difference method" (IPCC, 2006), is widely accepted for the estimation of stock changes using permanent NFI plot remeasurements. However, there is the potential for inconsistencies in stock transfers due to differences in sample protocols and unaccounted transfers (Domke et al., 2014; McRoberts et al., 2005; Woodall et al., 2019). Conceptually, the transitions of stems between various states and pools have been extensively documented (Garbarino et al., 2015; Harmon et al., 1986; Vanderwel et al., 2006). Woods et al (2021) examined the successional dynamics of old-growth forests around the world by using various long-term, local-scale datasets. While informative, their results were limited by inconsistencies across inventory protocols. Many studies specifically using FIA data have produced examinations of mortality trends (Dietze & Moorcroft, 2011; Lines et al., 2010), downed dead wood decomposition (Russell et al., 2013, 2014), drivers of standing dead wood fall rates (Oberle et al., 2018), and shifts in community composition (Knott et al., 2022). Yet, there remains limited work using FIA remeasurement data to examine long-term, longitudinal trends of holistic tree demographics.

Within FIA, the frequency of remeasurements differs within the eastern and western portions of the contiguous USA (primarily divided by the Mississippi River), i.e., 5- or 7-year vs. 10-year remeasurements, respectively. Thus, with the standardization of remeasurement protocols roughly two decades ago (McRoberts et al., 2005), areas within the eastern USA provide the best opportunity to quantify long-term demographics using FIA data over relatively short remeasurement periods. Moreover, with the contiguous USA spanning the temperate forest biome, the southern USA offers the best potential to observe changes in demographic rates over two decades given the warm, humid, and disturbance-prone nature of the region. East Texas has up to five consecutive measurements of plots and, although east Texas is a relatively small region of the USA, it provides an ideal case study for examining demographic trends for several reasons. Specifically, forests in east Texas are very diverse in terms of species (Marks & Harcombe, 1975), it is a highly productive region with intensively managed plantations and many unmanaged areas (Fox et al., 2007; Edgar & Zehnder, 2017), disturbances are frequent ranging from small to large scale (Edgar et al., 2019; Glitzenstein et al., 1986;

Hoerling et al., 2013), and the warm, humid climate causes a rapid change of both live and dead stems (Edgar et al., 2019; Klockow et al., 2018). Moreover, noteworthy disturbances occurred over the annual inventory which began in 2001 including five major hurricanes, i.e., Rita in 2005, Ike in 2008, Harvey in 2017, and Laura/Delta in 2020 (among other smaller hurricanes/tropical storms), and the worst drought ever recorded in Texas from September 2010 to October 2011 which affected the entire state (Hoerling et al., 2013; Nielsen-Gammon, 2012). These frequent, large-scale disturbances provide important context for examining demographic trends across FIA's large-scale inventory network and how they potentially drive resource changes in forest ecosystems.

Forest resource reporting techniques primarily focus on the two most recent measurements for understanding forest resource changes and may not realize more nuanced dynamics achieved by observing successive field plot measurements. By leveraging two decades of remeasurements, we intend to quantify live-dead wood demographics and, ultimately, better inform estimates and fluxes of stems in forest ecosystems. Our overall objective is to identify opportunities and gaps in tracking the demographics and dynamics of stems over multiple remeasurements within the FIA inventory framework using east Texas as a pilot study. Specifically, we will (1) examine growth and mortality rates of live stems over multiple remeasurements occurring between 2001 and 2021, (2) identify the long-term dynamics of standing stem removals and transitions to the downed dead wood pool, and (3) describe the nuances and implications of these processes within the context of the FIA inventory framework and associated forest dynamics assessments.

Materials and methods

Study area

East Texas has a sub-humid climate, receiving between 990 and 1600 mm of annual rainfall and warm annual temperatures between 16 and 23 °C (USDA NRCS, 2006). Given this, woody stem transitions (e.g., decomposition of dead wood) in warm, humid east Texas can occur rapidly (Klockow et al., 2018; Oberle et al., 2018; Zhang et al., 2016). In addition, east Texas experiences frequent, diverse, and large-scale disturbances including wildfires, drought, insect outbreaks, hurricanes, and wind events, as well as intensive management practices (e.g., plantations, regular prescribed fires) (Edgar & Zehnder, 2017; Edgar et al., 2019). These characteristics will allow us to obtain a comprehensive assessment of individual tree demography as trees and plots experience rapid change via climate, disturbance, and management.

East Texas represents the western extent of West Gulf Coastal Plain forests and is a highly productive and species-diverse region heavily dominated by pine, similar to much of the southeastern USA. Loblolly pine (*Pinus taeda* L.) is the most dominant softwood species occurring in both intensively managed plantations and naturally regenerated conditions (Edgar & Zehnder, 2017; Klockow et al., 2020). Hardwood species consist primarily of oaks, with the dominant oak species being post oak (*Quercus stellata* Wangenh.) and water oak (*Quercus nigra* L.) which generally occur in drier, upland conditions and moist, bottomland conditions, respectively (Burns & Honkala, 1990). Here, we focus on these four groups (i.e., planted loblolly, naturally regenerated loblolly, post oak, and water oak) given their prevalence throughout the region. Also, they provide an opportunity to examine trends in both softwoods and hardwoods, managed and unmanaged (i.e., loblolly pine), and differing physiographic preferences (i.e., post oak and water oak).

Data

The FIA program collects field data from a network of spatially balanced inventory plots laid out across the country (Bechtold & Patterson, 2005). A spatially even subset of these plots, termed a panel, is sampled annually, providing a statistically independent sample across the spatial extent of a given state or region every year. Panels of plots are remeasured at 5-, 7-, or 10-year intervals (or "remeasurement periods") with all plots in a region measured over this interval, termed a measurement "cycle." In east Texas, the annual inventory was initiated in 2001. The initial collection of these annual panel plot data was over a 3-year "accelerated" measurement cycle (i.e., one-third of plots measured annually from 2001 to 2003) with 5-year measurement cycles starting in 2004 (i.e.,

one-fifth of plots measured annually). Currently, all plots in east Texas have four total measurements and approximately 60% have five total measurements. Measurement cycles include 2001–2003, 2004–2008, 2009–2013, 2014–2018, and 2019–2021 (i.e., three panels of data available in the most recent cycle). We leverage these two decades of plot measurements to assess our objectives.

Plot layout consists of four total subplots, one central subplot and three additional subplots located at azimuths of 0, 120, and 240 relative to the central subplot with subplot centers 36.6 m apart. Each subplot comprises approximately 168.2 m^2 for a full plot total of approximately 672.8 m^2 and represents approximately 2400 ha on the landscape. Standing woody stems are classed into saplings (i.e., 2.54 cm ≤ DBH < 12.7 cm) and trees (i.e., DBH ≥ 12.7 cm). Trees are tallied on all subplots while saplings are tallied on four microplots, comprising 13.5 m^2 in each subplot for a total of 54 m^2 in each full plot. Individual saplings/trees are classified as either live or dead at each measurement, where stems are alive if they have any living parts (leaves, buds, cambium) at or above the point of diameter measurement, and this includes temporarily defoliated saplings/trees. Dead stems are further classified as standing or not (i.e., downed dead wood), where standing dead stems are at least 1.4 m tall and lean less than 45° from vertical. Standing dead stems continue to be measured at each plot visit until they no longer meet the criteria of a standing dead tree at which point, they are measured once more, classified as not standing dead, and no longer measured at subsequent plot visits. Live or dead trees that have been harvested (i.e., cut and removed from the plot) are classified as removed at the first plot visit after harvest. Even though the stem is no longer present, the stump of the previously live or dead tree remains and provides the evidence that the stem was harvested.

Measurements are collected across multiple levels including plots, conditions (i.e., mapped domains on plots), and trees. At least two types of conditions are identified on plots, forested or non-forested conditions, with forested conditions defined as > 10% tree canopy cover. For our purposes, data from the FIA database for east Texas were joined from the plot, condition, and tree tables to link individual plots and trees across all available measurements. Any individual condition (e.g., forest type or ownership class) cannot always be tracked over multiple remeasurements given their temporally dynamic nature. However, the conditions on an individual plot and associated with any particular tree can be tracked. Once plot, condition, and tree data were merged and linked across cycles, we identified those plots that were measured continuously through all available measurements. Thus, we have a common and consistent base of plots and trees for comparison across measurement cycles. From these plots, our sample included stems that existed in the first measurement cycle and those that appeared in any subsequent measurement cycle (i.e., ingrowth) as long as they had at least two consecutive measurements. We focus specifically on measurements of individual stems (as opposed to plot-level estimates, e.g., the number of stems or biomass or carbon per hectare within east Texas) as our goal was to understand demographics at a base level and how well the US NFI captures those demographics using east Texas as a pilot study region.

Demographic metrics and analyses

We examined four different categories of demographics, growth rates, mortality rates, stem-fall rates, and removal rates, in saplings and trees and subsequent transition rates out of the sample, i.e., to downed dead wood or litter/forest floor which encompass different measurement protocols that no longer track previously standing stems. We calculated demographic metrics separately for three different size classes including saplings 12.7 cm > DBH ≥ 2.54; small trees 25.4 cm > DBH ≥ 12.7 cm; and large trees DBH ≥ 25.4 cm. This provides consistency for comparison of growth, mortality, stem-fall rates, and removals.

Growth rates

For relative growth rate, we selected all remeasured, live stems that were ≥ 2.54 cm DBH in forested conditions on continuously measured plots. Live refers to the stem being alive at the initial measurement and at the next measurement of each remeasurement period; thus, growths from stems that were alive at the initial measurement and dead at the remeasurement were not included in growth rate calculations. Relative growth rates were calculated for each stem over each remeasurement period using the formula for mean

annual growth rate based on basal area, also referred to as periodic relative increment (Hunt, 1982; Pommerening & Muszta, 2016; Pile Knapp et al., 2021):

$$G_r = \frac{\left(\ln BA_2 - \ln BA_1\right)}{t}, \quad (1)$$

where G_r is the relative growth rate; BA_1 and BA_2 are the basal area (cross-sectional stem area at DBH) of a stem at initial measurement (BA_1) and remeasurement (BA_2), respectively; and t is the remeasurement period in years between initial measurement and remeasurement. This metric provides a standardized measure of growth for trees of various sizes (Pile Knapp et al., 2021) and remeasured over varying time intervals. Additionally, the three size classes of stems were analyzed separately to account for growth differences based on stem size since basal area growth tends to decrease as trees get larger (Pommerening & Muszta, 2016; Pile Knapp et al., 2021). Relative growth rates were averaged across all stems by size class and plotted with standard errors for each measurement cycle.

Mortality rates

For mortality rate calculations, we selected all remeasured stems, ≥ 2.54 cm DBH in forested conditions and on continuously measured plots. Stems were alive at the initial measurement and alive or dead at the next measurement for each remeasurement period. All harvested stems were excluded from these analyses, i.e., mortality rates do not include stems that were cut. Note that stems that were alive at initial measurement, died, and were subsequently removed before remeasurement were not able to be determined and are not counted in mortality rate calculations. These stems are instead included in removal rates further below. Annual mortality rates were calculated using the formula from Sheil et al (1995):

$$M = 1 - \left[1 - \frac{N_{t,M}}{N_{o,M}}\right]^{\frac{1}{t}}, \quad (2)$$

where M is the annual mortality rate, $N_{o,M}$ is the number of live trees in an initial measurement for mortality rate calculations, $N_{t,M}$ is the number of trees that died between each remeasurement period for mortality rate calculations, and t is the remeasurement period in years. The remeasurement period for plots in east Texas can vary approximately by ± 1 year due to logistics, access, etc. Given this, we calculated a weighted average of mortality rates as in Woods et al. (2021) to account for this variation in the remeasurement period. Specifically, we rounded remeasurement period years from tenths of a year (i.e., as reported in FIA's public database) to the nearest whole integer to provide larger sample sizes for calculation of weights (e.g., one stem for each of $t = 5.1$, 5.4, and 5.3 was modified to be three stems with $t = 5$). Next, we calculated mortality rates for each group of trees corresponding to each adjusted remeasurement period, calculated weights based on the number of stems for each adjusted remeasurement period (i.e., proportion of all stems), and calculated a weighted average of mortality rates for each full measurement cycle. Although the nature of the mortality formula and data did not allow for calculations of error (i.e., standard deviation or 95% confidence interval), we are able to report trends, consistency, and order across multiple measurement cycles. The three size classes of stems were analyzed separately to account for differences in mortality rates since these can vary by size (i.e., greater in smaller and often larger stems; Lines et al. (2010), Dietze and Moorcroft (2011), Klockow et al. (2020)).

Stem-fall rates

Transitions of stems from standing to downed are not often reported or studied using remeasurements. Conveniently, however, they represent a similar binary process as mortality and removals in that a stem is either standing (i.e., > 1.4 m tall and leaning < 45° from vertical) or downed (i.e., > 1.4 m tall and leaning ≥ 45° from vertical) after a period of time. We focus on two distinct processes inherent in stem-fall calculations for remeasured inventory plots: (1) trees that were alive at initial measurement and standing dead or downed at remeasurement, termed "tree-fall," and (2) trees that were standing dead at initial measurement and standing dead or downed at remeasurement, termed "snag-fall." For stem-fall rate calculations, we selected all remeasured stems (≥ 2.54 cm DBH for tree-fall and ≥ 12.7 cm DBH for snag-fall), in forested conditions and consistently measured plots. The standing dead status of saplings was not recorded until the inventory year 2016. Given this, there are no

or very minimal remeasurements of saplings with standing dead status codes available and we are thus unable to calculate snag-fall rates of saplings. All harvested stems were excluded from these analyses, i.e., stem-fall rates do not include stems that were cut. Annual stem-fall rates were calculated using the same formula as for mortality and removals (Sheil et al., 1995):

$$F = 1 - \left[1 - \frac{N_{t,F}}{N_{o,F}}\right]^{\frac{1}{t}}, \quad (3)$$

where F is the annual stem-fall rate, $N_{o,F}$ is the number of standing trees at initial measurement for stem-fall calculations, $N_{t,F}$ is the number of trees that fell between each remeasurement period for stem-fall calculations, and t is the remeasurement period in years. As with mortality and removal rates, we rounded remeasurement period years from tenths of a year to the nearest whole integer for the same reasons. Next, we calculated stem-fall rates for each group of trees corresponding to each adjusted remeasurement period, calculated weights based on the number of stems for each adjusted remeasurement period (i.e., proportion of all stems), and calculated a weighted average of fall rates for each measurement cycle. These methods were applied to both the tree-fall and snag-fall groups separately. As noted for mortality and removal rates, although the nature of the mortality formula and data did not allow for calculations of error (i.e., standard deviation or 95% confidence interval), we are able to report trends, consistency, and order across multiple measurement cycles. The two size classes of tree stems were analyzed separately adding greater resolution to stem-fall rates as these can vary by tree size (Oberle et al., 2018; Vanderwel et al., 2006).

Removal rates

For removal rate calculations, we selected all remeasured stems, ≥ 2.54 cm DBH in forested conditions and on continuously measured plots. Stems were uncut (i.e., alive or dead) at the initial measurement and cut (i.e., removed) or uncut at the next measurement. Annual removal rates were calculated using the formula for mortality from Sheil et al (1995):

$$R = 1 - \left[1 - \frac{N_{t,R}}{N_{o,R}}\right]^{\frac{1}{t}}, \quad (4)$$

where R is the annual removal rate, $N_{o,R}$ is the number of uncut trees in an initial measurement for removal rate calculations, $N_{t,R}$ is the number of trees that were cut between each remeasurement period for removal rate calculations, and t is the remeasurement period in years. As with mortality and stem-fall rates, we rounded remeasurement period years from tenths of a year to the nearest whole integer for the same reasons. Next, we calculated removal rates for each group of trees corresponding to each adjusted remeasurement period, calculated weights based on the number of stems for each adjusted remeasurement period (i.e., proportion of all stems), and calculated a weighted average of removal rates for each measurement cycle. As noted for mortality and stem-fall rates, although the nature of the mortality formula and data did not allow for calculations of error (i.e., standard deviation or 95% confidence interval), we are able to report trends, consistency, and order across multiple measurement cycles. The three size classes of stems were analyzed separately adding greater resolution to removal rates.

Results

Data used to determine demographic rates are summarized by each demographic category and presented in tables. Summarized data include number of plots, median DBH, maximum DBH, and number of stems by species group, measurement cycle, and size class. Summaries for growth rate data are in Table 1, mortality rate data in Table 2, tree-fall rate data in Table 3, snag-fall rate data in Table 4, and removal rate data in Table 5.

Growth rates

Growth rate trends by size classes were consistent across the species groups examined and were consistently highest in saplings, followed by small trees, and lowest in large trees (Fig. 1, Table S1). Growth rates were highest in planted loblolly saplings varying between approximately 7.5 and 1%/year, while growth rates varied between approximately 1 and

Table 1 Growth summary data (minimum DBH is 2.54 cm for all groups). DBH corresponds to the initial measurement for each measurement cycle

Species group	Measurement cycle (initial)	Measurement cycle (remeasure)	Plots (#)	Saplings (#)	Sapling DBH median (cm)	Small trees (#)	Small tree DBH median (cm)	Large trees (#)	Large tree DBH median (cm)	Large tree DBH maximum (cm)
All species	2001–2003	2004–2008	1705	9034	4.8	20,759	16.8	7941	33.0	122.2
	2004–2008	2009–2013	1725	7899	5.1	20,077	16.8	7323	33.0	147.8
	2009–2013	2014–2018	1711	7401	5.1	20,128	17.0	7508	32.5	142.0
	2014–2018	2019–2021	1008	4172	5.3	12,911	17.3	5209	32.3	124.2
Loblolly (planted)	2001–2003	2004–2008	404	943	6.6	5705	17.0	868	29.0	70.1
	2004–2008	2009–2013	400	722	7.6	5708	16.8	796	29.0	68.1
	2009–2013	2014–2018	432	705	6.9	5859	17.0	1077	28.7	62.5
	2014–2018	2019–2021	252	373	7.9	3404	17.3	813	29.2	63.5
Loblolly (naturally regenerated)	2001–2003	2004–2008	719	1005	5.8	3162	17.0	1948	35.1	90.2
	2004–2008	2009–2013	732	733	6.6	3112	16.8	1917	35.1	91.9
	2009–2013	2014–2018	758	706	5.3	3465	17.8	2039	35.1	93.0
	2014–2018	2019–2021	471	352	7.1	2476	18.0	1516	34.0	89.7
Post oak	2001–2003	2004–2008	444	173	5.6	1105	18.0	577	32.3	74.7
	2004–2008	2009–2013	412	151	5.6	925	18.3	529	32.5	76.2
	2009–2013	2014–2018	394	133	5.8	758	18.5	490	32.8	76.7
	2014–2018	2019–2021	223	87	6.4	426	18.5	309	33.3	86.4
Water oak	2001–2003	2004–2008	658	658	4.3	767	17.0	446	35.2	113.5
	2004–2008	2009–2013	636	589	4.6	821	16.3	424	34.8	90.4
	2009–2013	2014–2018	662	532	5.1	956	16.5	432	35.8	86.1
	2014–2018	2019–2021	414	306	5.2	726	16.5	296	34.2	85.1

Table 2 Mortality summary data (minimum DBH is 2.54 cm for all groups). DBH corresponds to the initial measurement for each measurement cycle. Number of stems is the number alive in the initial measurement cycle

Species group	Measurement cycle (initial)	Measurement cycle (remeasure)	Plots (#)	Saplings (#)	Sapling DBH median (cm)	Small trees (#)	Small tree DBH median (cm)	Large trees (#)	Large tree DBH median (cm)	Large tree DBH maximum (cm)
All species	2001–2003	2004–2008	1708	10,061	4.6	21,512	16.8	8233	33.0	122.2
	2004–2008	2009–2013	1730	9725	4.8	21,856	16.8	8181	33.0	147.8
	2009–2013	2014–2018	1728	9445	4.8	22,489	17.0	8326	32.8	142.0
	2014–2018	2019–2021	1013	5194	5.1	13,675	17.3	5370	32.3	124.2
Loblolly (planted)	2001–2003	2004–2008	404	1077	6.1	5803	17.0	880	29.0	70.1
	2004–2008	2009–2013	402	880	7.1	5911	16.8	826	29.0	68.1
	2009–2013	2014–2018	441	859	6.4	6152	17.0	1105	28.7	62.5
	2014–2018	2019–2021	252	511	6.9	3527	17.3	819	29.2	63.5
Loblolly (naturally regenerated)	2001–2003	2004–2008	732	1241	5.3	3250	17.0	1990	35.2	95.5
	2004–2008	2009–2013	745	1038	5.8	3344	16.8	2056	35.1	92.0
	2009–2013	2014–2018	774	964	5.1	3777	17.5	2191	35.1	93.0
	2014–2018	2019–2021	477	534	5.8	2582	18.0	1548	34.0	93.7
Post oak	2001–2003	2004–2008	454	194	5.3	1151	18.0	588	32.5	74.7
	2004–2008	2009–2013	428	181	5.3	1007	18.3	565	32.8	76.2
	2009–2013	2014–2018	419	170	5.3	871	18.3	551	32.8	77.5
	2014–2018	2019–2021	230	102	6.1	444	18.5	316	33.3	86.4
Water oak	2001–2003	2004–2008	671	703	4.3	787	17.0	469	35.3	113.5
	2004–2008	2009–2013	680	719	4.3	909	16.3	494	35.1	114.8
	2009–2013	2014–2018	708	706	4.6	1110	16.5	494	35.8	93.5
	2014–2018	2019–2021	423	365	4.8	747	16.5	304	34.5	85.1

Table 3 Tree-fall summary data (minimum DBH is 2.54 cm for all groups). DBH corresponds to the initial measurement for each measurement cycle. Number of stems is the number alive in the initial measurement cycle and which subsequently died at remeasurement (i.e., none was still alive at remeasurement)

Species group	Measurement cycle (initial)	Measurement cycle (remeasure)	Plots (#)	Saplings (#)	Sapling DBH median (cm)	Small trees (#)	Small tree DBH median (cm)	Large trees (#)	Large tree DBH median (cm)	Large tree DBH maximum (cm)
All species	2001–2003	2004–2008	793	1027	3.8	753	16.8	292	34.4	97.3
	2004–2008	2009–2013	1213	1826	4.1	1779	16.8	858	34.5	117.9
	2009–2013	2014–2018	1285	2044	4.3	2361	16.5	818	34.0	97.8
	2014–2018	2019–2021	636	1022	4.3	764	15.7	161	32.8	114.3
Loblolly (planted)	2001–2003	2004–2008	102	134	4.6	98	17.1	12	29.2	35.3
	2004–2008	2009–2013	156	158	5.3	203	16.5	30	29.1	45.2
	2009–2013	2014–2018	164	154	5.0	293	17.0	28	29.6	43.4
	2014–2018	2019–2021	89	138	4.8	123	15.7	6	27.3	31.8
Loblolly (naturally–regenerated)	2001–2003	2004–2008	161	236	3.9	88	16.5	42	38.5	95.5
	2004–2008	2009–2013	286	305	4.6	232	16.0	139	38.4	89.4
	2009–2013	2014–2018	260	258	4.1	312	16.5	152	34.3	68.6
	2014–2018	2019–2021	133	182	4.6	106	14.7	32	32.9	93.7
Post oak	2001–2003	2004–2008	54	21	4.3	46	16.5	11	36.1	49.3
	2004–2008	2009–2013	91	30	5.0	82	17.0	36	33.1	72.4
	2009–2013	2014–2018	106	37	3.8	113	17.3	61	33.5	77.5
	2014–2018	2019–2021	32	15	5.3	18	17.9	7	32.5	66.3
Water oak	2001–2003	2004–2008	63	45	3.8	20	17.7	23	38.6	85.9
	2004–2008	2009–2013	177	130	3.8	88	17.8	70	37.6	114.8
	2009–2013	2014–2018	208	174	3.8	154	16.4	62	36.1	93.5
	2014–2018	2019–2021	62	59	3.8	21	16.8	8	40.1	49.8

Table 4 Snag-fall summary data (minimum DBH is 12.7 cm for all groups, i.e., no saplings). DBH corresponds to the initial measurement for each measurement cycle. Number of stems is the number of standing dead in the initial measurement cycle

Species group	Measurement cycle (initial)	Measurement cycle (remeasure)	Plots (#)	Saplings (#)	Sapling DBH median (cm)	Small trees (#)	Small tree DBH median (cm)	Large trees (#)	Large tree DBH median (cm)	Large tree DBH maximum (cm)
All species	2001–2003	2004–2008	714	-	-	1047	17.5	462	33.3	90.2
	2004–2008	2009–2013	633	-	-	746	17.5	389	33.5	95.3
	2009–2013	2014–2018	789	-	-	1094	17.0	641	34.8	114.6
	2014–2018	2019–2021	491	-	-	666	17.8	425	34.8	94.0
Loblolly (planted)	2001–2003	2004–2008	57	-	-	157	17.8	17	28.4	38.6
	2004–2008	2009–2013	54	-	-	72	18.2	11	28.2	37.1
	2009–2013	2014–2018	73	-	-	96	17.0	29	30.5	40.6
	2014–2018	2019–2021	37	-	-	42	18.8	12	30.5	53.8
Loblolly (naturally regenerated)	2001–2003	2004–2008	150	-	-	179	17.0	77	32.0	65.8
	2004–2008	2009–2013	125	-	-	110	17.3	50	33.9	95.3
	2009–2013	2014–2018	144	-	-	135	16.0	96	40.4	90.4
	2014–2018	2019–2021	87	-	-	85	18.8	70	36.4	81.3
Post oak	2001–2003	2004–2008	94	-	-	99	17.8	49	34.8	78.7
	2004–2008	2009–2013	85	-	-	90	17.5	46	37.5	77.7
	2009–2013	2014–2018	83	-	-	101	18.0	44	31.9	74.7
	2014–2018	2019–2021	57	-	-	70	17.8	57	32.0	68.3
Water oak	2001–2003	2004–2008	51	-	-	41	17.3	29	35.8	90.2
	2004–2008	2009–2013	42	-	-	22	17.8	28	38.2	87.1
	2009–2013	2014–2018	77	-	-	69	16.3	44	42.5	113.5
	2014–2018	2019–2021	63	-	-	47	16.5	38	37.3	84.3

Table 5 Removal summary data (minimum DBH is 2.54 cm for all groups). DBH corresponds to the initial measurement for each measurement cycle. Number of stems is the total number of stems in the initial measurement cycle

Species group	Measurement cycle (initial)	Measurement cycle (remeasure)	Plots (#)	Saplings (#)	Sapling DBH median (cm)	Small trees (#)	Small tree DBH median (cm)	Large trees (#)	Large tree DBH median (cm)	Large tree DBH maximum (cm)
All species	2001–2003	2004–2008	1760	11,254	4.6	26,038	17.0	9847	32.8	122.2
	2004–2008	2009–2013	1796	11,275	4.8	26,617	16.8	9832	32.8	147.8
	2009–2013	2014–2018	1800	10,892	4.8	28,581	17.0	10,177	32.5	142.0
	2014–2018	2019–2021	1051	5791	5.1	16,529	17.3	6377	32.3	124.2
Loblolly (planted)	2001–2003	2004–2008	432	1225	6.4	7924	17.0	1216	29.0	70.1
	2004–2008	2009–2013	443	1045	7.1	8046	16.8	1297	29.2	68.1
	2009–2013	2014–2018	491	1102	6.6	9521	17.3	1723	29.0	62.5
	2014–2018	2019–2021	276	607	7.1	4946	17.3	1117	29.2	63.5
Loblolly (naturally regenerated)	2001–2003	2004–2008	796	1357	5.3	3901	17.0	2461	34.8	95.5
	2004–2008	2009–2013	822	1231	5.8	4145	17.0	2447	34.8	95.3
	2009–2013	2014–2018	824	1111	5.3	4521	17.5	2562	35.3	93.0
	2014–2018	2019–2021	513	597	5.8	3027	18.0	1771	34.3	93.7
Post oak	2001–2003	2004–2008	499	219	5.3	1341	18.0	685	32.5	78.7
	2004–2008	2009–2013	474	214	5.3	1217	18.3	660	32.8	77.7
	2009–2013	2014–2018	455	182	5.3	1016	18.3	622	32.6	77.5
	2014–2018	2019–2021	255	104	6.1	531	18.5	382	32.9	86.4
Water oak	2001–2003	2004–2008	720	787	4.1	908	16.8	519	35.3	113.5
	2004–2008	2009–2013	749	847	4.3	1015	16.3	561	35.1	114.8
	2009–2013	2014–2018	791	819	4.6	1296	16.5	571	36.1	113.5
	2014–2018	2019–2021	458	401	4.8	836	16.5	353	35.1	85.1

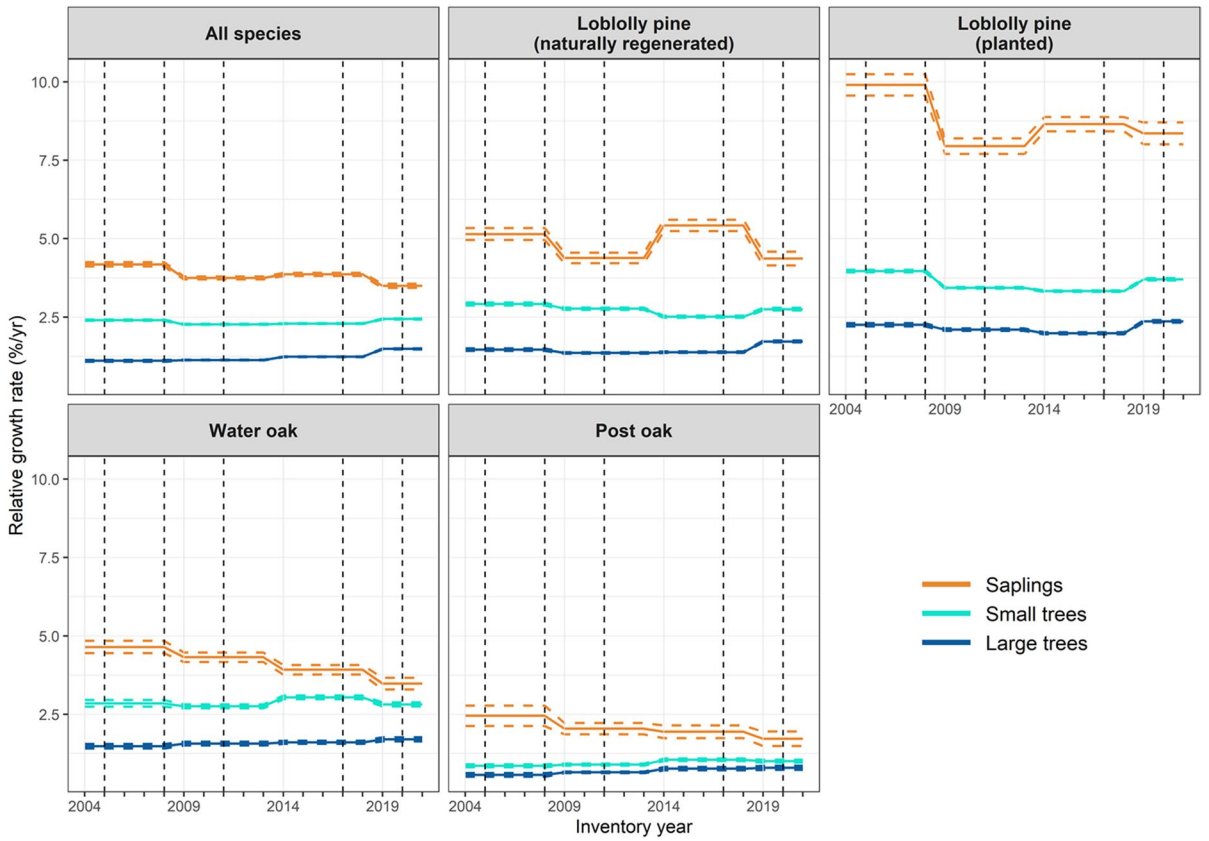

Fig. 1 Mean relative basal area growth rates with standard errors (colored dashed lines) across all cycles and all species groups examined by size classes on forested land. Inventory years shown indicate the starting year of measurement cycles (i.e., 2004–2008, 2009–2013, 2014–2018, 2019–2021). Vertical black dashed lines at 2005, 2008, 2011, 2017, and 2020 highlight the timing of notable major disturbances including Hurricane Rita, Hurricane Ike, the 2011 drought, Hurricane Harvey, and Hurricanes Laura/Delta, respectively

5.5%/year for all other species groups and size classes. Growth was relatively stable for all species, primarily small and large trees across all measurement cycles. Growth declined for all groups and size classes during the measurement cycle encompassing the 2011 drought (i.e., 2009–2013) with the exception of small post oak and large water and post oak trees. Post oak tree growth remained stable during the remeasurement periods encompassing the 2011 drought and increased slightly in each measurement cycle thereafter. Trees in both loblolly pine groups showed decreased growth in the measurement cycle after the drought (i.e., 2014–2018). Interestingly, growth rates of pine saplings increased while oak sapling growth rates continued to decline after the drought, showing opposite trends. Also, growth rates of oak trees continued increasing after the drought while oak sapling growth rates continued declining after the drought.

Mortality rates

Saplings consistently showed the highest mortality rates across all species and measurement cycles, with the highest rates in naturally regenerated loblolly pine (i.e., approximately 5–8%, Fig. 2, Table S1). Mortality rates varied the least for both small and large planted loblolly pine trees (i.e., approximately 1% or less) and varied the most for both small and large water oak trees (i.e., approximately < 1% up to 3%). Notably, the impact of the 2011 drought on stem mortality is clearly evident once it became part of the measurement cycle (i.e.,

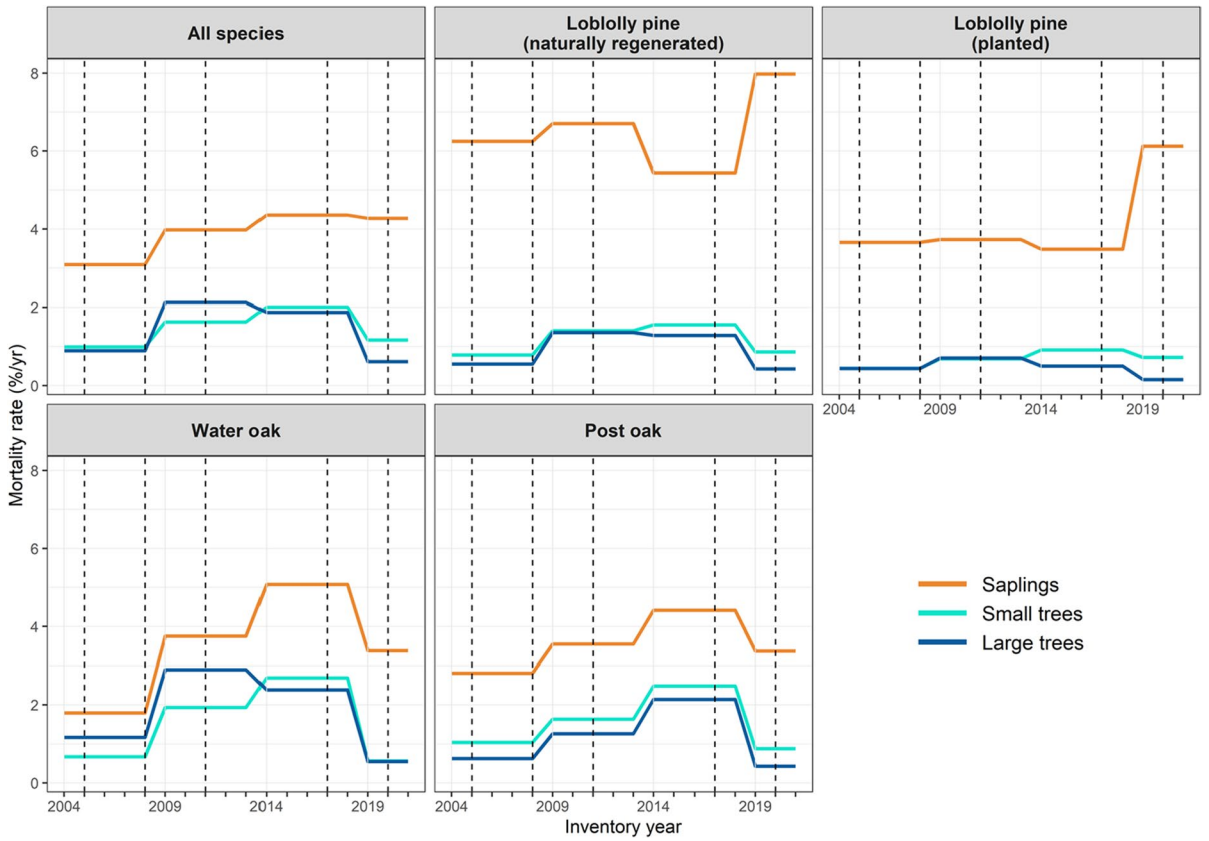

Fig. 2 Mortality rates across all cycles and all species groups examined by size classes on forested land, excluding harvested stems. Inventory years shown indicate the starting year of measurement cycles (i.e., 2004–2008, 2009–2013, 2014–2018, 2019–2021). Vertical black dashed lines at 2005, 2008, 2011, 2017, and 2020 highlight the timing of notable major disturbances including Hurricane Rita, Hurricane Ike, the 2011 drought, Hurricane Harvey, and Hurricanes Laura/Delta, respectively. Note that the nature of the mortality rate formula did not allow for the calculation of error

2009–2013) with mortality rates increasing for all species groups and size classes. Mortality rates continued increasing into the next measurement cycle following the drought (i.e., 2014–2018) for smaller pine and water oak trees and for all post oak trees. Mortality rates stabilized or decreased for larger pine and water oak trees in the measurement cycle after the drought. Of the large tree size group, only post oak trees showed a continued increase in mortality rates in the measurement cycle after the drought (i.e., 2014–2018). Interestingly, all species of trees showed decreased mortality rates and reordering by size class in the most recent measurement cycle (i.e., 2019–2021), approximately reaching rates and orders in the measurement cycle preceding the drought (i.e., 2004–2008).

Stem-fall rates

Tree-fall rates were consistently highest in saplings. Sapling rates were 100% in the first measurement cycle and dropped substantially to 25–50% by the most recent measurement cycle (Fig. 3, Table S1). Tree-sized stems remained more stable than saplings across all measurement cycles and were less than 25% for all measurement cycles. Tree-fall rates for small and large trees were variable across measurement cycles and species groups with large tree rates tending to be slightly lower than for small trees. Small pine tree-fall rates increased over time before decreasing in the most recent measurement cycle. Small water oak tree-fall rates generally increased over time while small post oak rates generally decreased over time. Large tree tree-fall rates varied most for planted

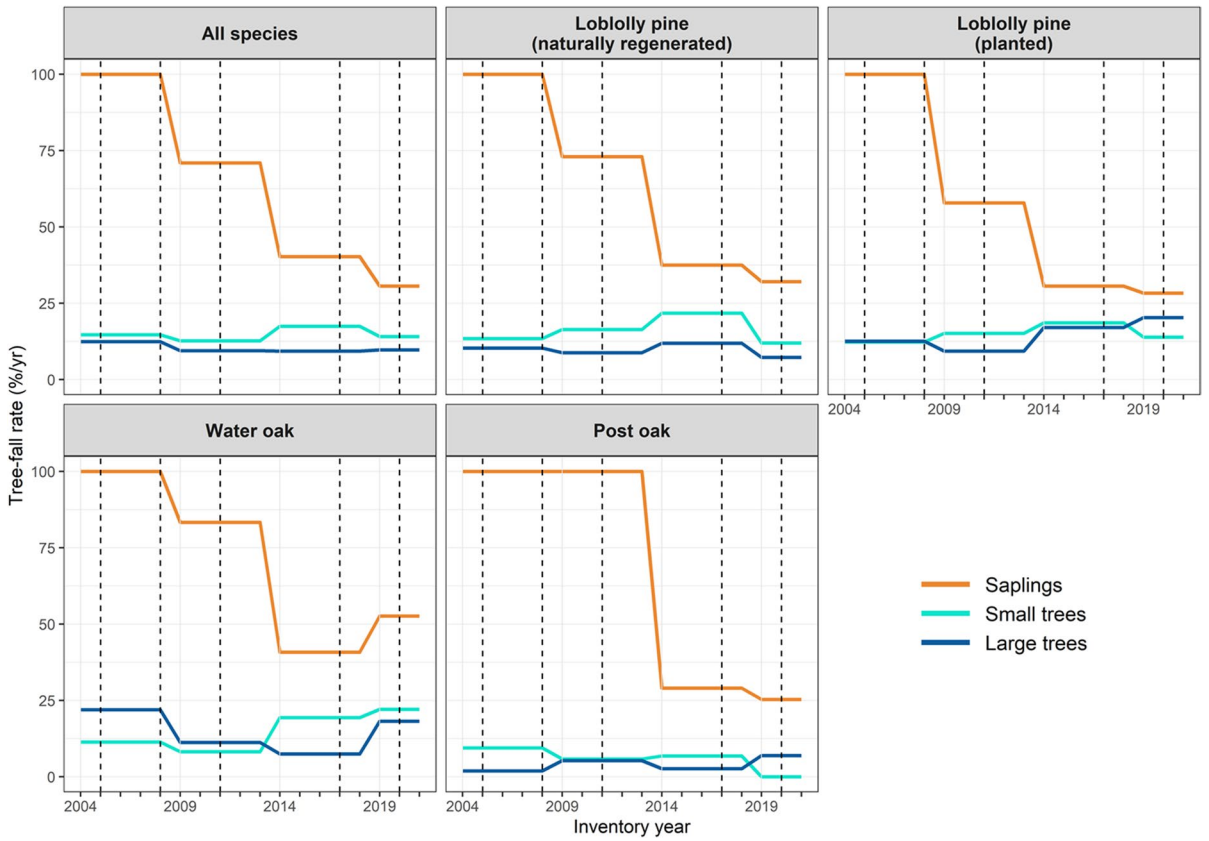

Fig. 3 Tree-fall rates across all cycles and all species groups examined by size classes on forested land, excluding harvested stems. Inventory years shown indicate the starting year of measurement cycles (i.e., 2004–2008, 2009–2013, 2014–2018, 2019–2021). Vertical black dashed lines at 2005, 2008, 2011, 2017, and 2020 highlight the timing of notable major disturbances including Hurricane Rita, Hurricane Ike, the 2011 drought, Hurricane Harvey, and Hurricanes Laura/Delta, respectively. Note that tree-fall rates are relative to only those live trees that died between each measurement and not all live trees, i.e., of all trees that died in each measurement cycle, fall rate is the percentage that fell. Note that the nature of the fall rate formula did not allow for the calculation of error

loblolly and water oak but remained more stable for large naturally regenerated pines and post oaks.

Snag-fall rates showed more variability over time and across species than for small and large tree tree-fall rates (Fig. 4, Table S1). Snag-fall of planted loblolly appeared to occur in pulses over time with large planted loblolly snags having the highest snag-fall rates (i.e., approximately 75–100%) relative to the other species of large trees (i.e., approximately 10–40%) examined across all measurement cycles. Snag-fall rates increased substantially for smaller planted loblolly snags (i.e., from approximately 50 to 100%) in the most recent measurement cycle (i.e., 2019–2021). Similar to planted loblolly, water oak appears to have experienced pulses of snag-fall over the measurement cycles. Smaller water oak trees showed much greater snag-fall rates than larger water oak stems varying between approximately 60 and 100%.

Removal rates

Stem removal rates were relatively high for all species groups and size classes in the first measurement cycle ranging between 40 and 65% (Fig. 5, Table S1). Following this, removal rates declined sharply, to roughly between 30 and 40%, for all species and size classes with the exception of planted loblolly pine which declined even lower to between 20 and 30%. These rates remained low for the next

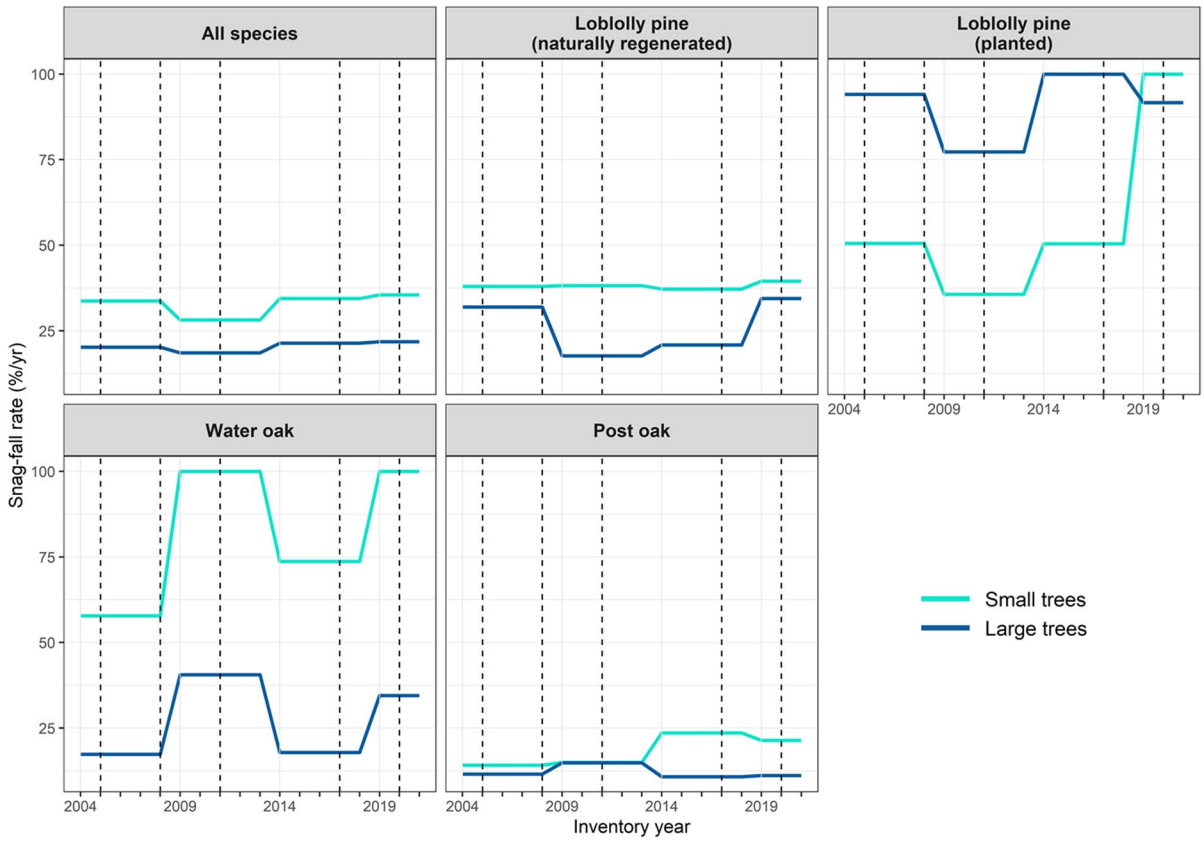

Fig. 4 Snag-fall rates across all cycles and all species groups examined by size classes on forested land, excluding harvested stems. Inventory years shown indicate the starting year of measurement cycles (i.e., 2004–2008, 2009–2013, 2014–2018, 2019–2021). Vertical black dashed lines at 2005, 2008, 2011, 2017, and 2020 highlight the timing of notable major disturbances including Hurricane Rita, Hurricane Ike, the 2011 drought, Hurricane Harvey, and Hurricanes Laura/Delta, respectively. Note that the standing dead status of saplings was not measured until 2016; thus, no repeat measurements of this pool are currently available. Note that the nature of the fall rate formula did not allow for the calculation of error

two measurement cycles, 2009–2013 and 2014–2018, before increasing again for all species and size classes but not to removal rates from the first measurement cycle 2004–2008. Across size classes, large trees tended to have the highest removal rates with the exception of planted loblolly pine, where large trees had the lowest removal rates. Saplings had the lowest removal rates for both oak groups while saplings had the highest removal rates for planted loblolly pine.

Discussion

Demographic rates represent the key metric describing resource changes in forested ecosystems. Estimates of volume, biomass, and carbon can be ascribed to any stem but it is the underlying demographic status change of individual stems as identified during field inventory that ultimately defines forest resource changes. Here, we move beyond calculating demographic rates from the two most recent points in time by focusing instead on the full suite of measurements available within the NFI of the USA, the FIA inventory, and specifically examine east Texas as a pilot study region. We focus on the main components of demography captured by FIA: growth, mortality, fall rates, and removals of saplings and trees.

Growth and mortality rates

Noteworthy growth and mortality dynamics emerged within the species groups examined around the time

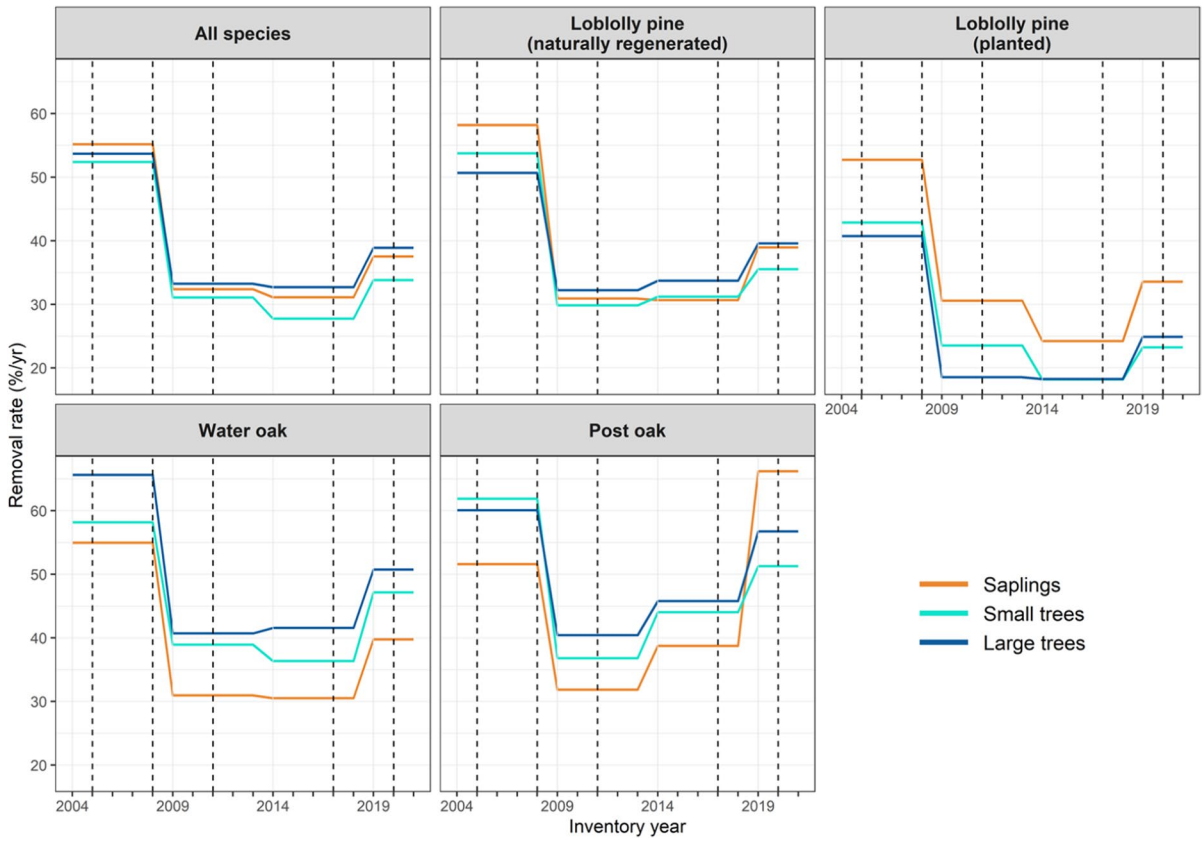

Fig. 5 Removal rates across all cycles and all species groups examined by size classes on forested land. Inventory years shown indicate the starting year of measurement cycles (i.e., 2004–2008, 2009–2013, 2014–2018, 2019–2021). Vertical black dashed lines at 2005, 2008, 2011, 2017, and 2020 highlight the timing of notable major disturbances including Hurricane Rita, Hurricane Ike, the 2011 drought, Hurricane Harvey, and Hurricanes Laura/Delta, respectively. Note that the nature of the removal rate formula did not allow for the calculation of error

of the 2011 drought and after. Specifically, growth declined for all groups and size classes in the measurement cycle encompassing the 2011 drought, i.e., 2009–2013, with the exception of large oak trees and small post oak trees. Notably, growth rates of oak trees increased in the measurement cycle after the drought suggesting fairly rapid recovery or sustained growth, while pine tree growth rates continued to decline in the measurement cycle after the drought suggesting lagged growth responses possibly related to internal hydraulic damage or reduced carbon stores (Anderegg et al., 2013; D'Amato et al., 2013; Berdanier & Clark, 2016, Cailleret et al., 2017). Regarding mortality, studies have highlighted the magnitude of this event primarily, and understandably, by capturing the major initial pulse of mortality (Moore et al., 2016; Schwantes et al., 2017), which increased substantially for all species when the drought began. However, as seen here and in similar studies, tree mortality responses to drought can vary widely and often have lagged mortality multiple years after the event has passed (Berdanier & Clark, 2016; Bigler et al., 2007; Cailleret et al., 2017; Klockow et al., 2018). Moreover, recent global studies have suggested that larger trees are most vulnerable to drought (Bennett et al., 2015). This was seen in part in our results particularly for large water oak stems. However, with the exception of post oak, larger trees appeared to recover in the measurement cycle immediately following the drought. All smaller trees and oak saplings showed increased mortality in the measurement cycle immediately following the drought, suggesting continued lagged mortality. Thus, while large stems may be vulnerable to initial drought

conditions, they may be capable of relatively quick recovery if secondary stress from pests and diseases remains limited while smaller stems may see continued, lagged mortality after the event has passed possibly due to more limited access to resources (e.g., shallower rooting depth, less leaf area). Interestingly, mortality rates of all trees returned to pre-drought levels starting in the most recent measurement cycle (i.e., 2019–2021) suggesting that mortality rates recovered to pre-drought levels by that time. These long-term trends shed light on the potential recovery ability of important species groups following water stress, which is largely predicted to become more common in the future (Allen et al., 2015).

These growth and mortality trends highlight the widespread and intense nature of the drought that impacted east Texas. They are generally consistent with species-specific strategies for coping with water stress. Although physiologic responses to water stress can vary among and within species, trees generally tend toward isohydric (i.e., maintaining relatively constant transpiration rates through stomatal regulation) or anisohydric (i.e., variable transpiration rates with less stomatal regulation) strategies (Adams et al., 2017; Anderegg et al., 2016; McDowell et al., 2008). Loblolly pines often show isohydric responses to water stress at the expense of growth via CO_2 uptake, i.e., leaf abscission and stomatal regulation (Maggard et al., 2016), evidenced here by the reduced growth around the time of the drought, whereas pine mortality remained relatively low (i.e., planted loblolly) or stable (i.e., naturally regenerated loblolly) compared to oaks and, in general, was likely mitigated by these physiologic responses and favorable growing conditions (e.g., reduced densities, favorable species mixes) (Bottero et al., 2017; Fox et al., 2007; Gleason et al., 2017; Klockow et al., 2020). Conversely, oaks tend to have an anisohydric response to water stress allowing them to remain productive while operating at an increased risk of embolism, i.e., less stomatal regulation (Pataki & Oren, 2003, Will et al., 2013; Rodriguez-Calcerrada et al., 2017), evidenced by the sustained growth rates during the drought. Notably, this likely resulted in rapid mortality of many oak stems (i.e., within a year or two) given the magnitude of the drought conditions and, for others, reduced tree vigor leading to lagged mortality via drought stress or other factors such as insects and diseases (Anderegg et al., 2015; Klockow et al., 2018). The factors and mechanisms affecting growth and ultimately leading to mortality of any particular stem, even in the context of such an extreme event, are influenced by biotic and abiotic factors at intra-tree, inter-tree, and site levels (Anderegg et al., 2013). By leveraging up to five measurements of stems across 20 years, we can begin to understand the range of variation growth and mortality rates which exhibits over time, particularly in relation to disturbance events.

Stem-fall rates

The transition from live to dead (i.e., tree mortality) is important in that a tree goes from a sink to a source of carbon. However, the transition from standing to downed provides further resolution into trees as a source of carbon. Specifically, once in contact with the ground, wood may become more susceptible to decay or prescribed fire/wildfire and, ultimately, the release of carbon to the atmosphere (Russell et al., 2015). The east Texas NFI data presented here show that tree-fall and snag-fall are both dynamic, commonly occurring processes in this warm, humid, disturbance-prone region. As with growth and mortality rates, stem size is an important factor in differentiating fall rates of both trees and snags (Moorman et al., 1999; Oberle et al., 2018; Vanderwel et al., 2006; Zarnoch et al., 2013). Tree-fall rates of saplings were far greater than for trees in the first two measurement cycles while small and large trees showed similar fall rates across all measurement cycles. The nuanced dynamics of fall rates for trees are difficult to attribute without further detailed analyses which is beyond the scope of this paper. For example, it is difficult to infer from our results how the 2011 drought may have impacted fall rates. It seems logical to suggest that absolute numbers of fallen stems have increased due to a larger pool of dead stems from the drought but whether fall rates have changed as a result of the drought is more challenging to deduce from our direct results. However, it seems plausible that the combinations of Hurricane Rita in 2005 and Hurricane Ike in 2008 were key drivers of sapling fall rates given the consistency and dramatic changes (i.e., decreases) for all species in the first two measurement cycles. Other studies have noted hurricane-driven stem-fall as an important process in this region (Harcombe & Marks, 1978; Harcombe et al., 2002, 2009). Moreover, the decomposer community in east Texas and

much of the southern USA includes termites which is another important driver of stem-fall rates (Zhang et al., 2016) in addition to the regional climate and frequent disturbances (e.g., hurricanes).

The means by which a stem transitions from standing to downed can occur via two primary processes, a live stem either falls and dies simultaneously or dies and remains standing for some time while undergoing decomposition and eventually falls (Harmon et al., 1986; Vanderwel et al., 2006). There is some overlap in these processes when considering inventory-based observation as some tree-fall stems may have died, remained standing for a short time, and then fallen all within a single remeasurement period. This is a caveat of any long-term inventory with annual or decadal remeasurement periods. However, the key factors which affect tree-fall are similar to those that affect snag-fall, e.g., stem size (Oberle et al., 2018; Vanderwel et al., 2006). The primary difference is that snags have undergone a known (i.e., measured) amount of decay. Regardless, few studies have examined these two processes (Klockow, 2019; Vanderwel et al., 2006) while others focused primarily on standing dead to downed fall rates (Oberle et al., 2018). In a region such as east Texas, these two processes can occur extensively and rapidly (Conner & Saenz, 2005; Putman et al., 2018). Accounting for both of these processes provides increased resolution and an important link between standing and downed tree habitat, fuel loads, and carbon pools.

Removal rates

It is important to note that the sample of stems we examined across measurement cycles was not static, meaning, we did not examine only those specific stems measured in the first measurement cycle through to the last measurement cycle. The number of stems we examined fluctuated at any measurement year as stems entered as ingrowth (i.e., meeting criteria of saplings or trees) and left as down dead (i.e., no longer measured). Thus, the high removal rates in the first measurement cycle are not indicating that all sampled stems were removed since more stems entered the sample as ingrowth at each subsequent measurement. Interestingly, removal rates for all species groups declined substantially during the measurement cycle containing the 2011 drought (i.e., 2009–2013). This follows in line with estimates of removed volume from the Texas A&M Forest Service which were 717 million ft^3 (approximately 20.3 million m^3) in 2010 and dropped to between 538.9 and 645.9 million ft^3 (approximately 15.3 to 18.3 million m^3) since (i.e., 2020 and 2011 estimates, respectively) (Edgar 2010, 2011; Brown et al., 2020). Presumably, removal rates were reduced given the substantial mortality incurred from the drought, which limited the number of growing stock stems available for harvest. However, as mortality rates declined and growth recovered in the two measurement cycles following the drought, removal rates also rebounded. While natural disturbances (e.g., drought, hurricanes, fire) are important drivers of removal rates, so are socioeconomic factors. Housing market and construction trends can be very important for assessing removal rates, especially in a region such as east Texas which has extensive numbers of pine plantations. Specifically, the housing market collapse in 2006 and the "Great Recession" around 2008 resulted in the worst economic conditions in the forest products industry in many decades (Woodall et al., 2011). Moreover, it remains to be seen in the coming inventory years how the COVID-19 pandemic may have influenced removal rates within FIA measurements. During the pandemic, demand for wood products increased as more people had time at home for renovations; at the same time, many forest products companies required loans to keep staff on payroll during quarantine periods with limited production (Russell, 2022; Zhang & Stottlemyer, 2021). However, since this socioeconomic information is not part of standard FIA measurements, we do not present it here. Timber products are tracked by a separate inventory within FIA, i.e., Timber Products Output (TPO), which surveys wood-using mills for volume and products (Coulston et al., 2018). While removal rates as presented here can be somewhat ambiguous for determining the fate of forest products, incorporating ancillary data from TPO and other sources can be important for continuing to track wood-based material and its carbon from harvested stems once it leaves the forest ecosystem.

Opportunities and gaps within the FIA framework

Temporal considerations

The effects of the 2011 drought are very evident in the growth and mortality trends for east Texas FIA

data. Unprecedented drought events like this are predicted to be more common in the near future (Allen et al., 2015). Importantly, multiple remeasurements here provide greater context to growth and mortality rates particularly when those measurements encompass a major disturbance event. Specifically, continued declines in growth and lagged mortality were evident, yet it appears these have started to return to pre-drought levels, suggesting recovery. However, the temporal resolution of these trends is limited by the specific inventory remeasurement period, i.e., 5 years in east Texas. This lag bias is well documented in FIA-related literature (Bechtold & Patterson, 2005). The panel system used by FIA overcomes this to an extent by offering annual, spatially balanced samples with better resolution compared to the full set of panels (Edgar et al., 2019). However, these annual measurements only offer static estimates for any given year as opposed to remeasurement of the same stems on an annual basis which would provide rates of change. This limited temporal resolution can make it difficult to ascertain short-term lagged impacts or distinguish between multiple disturbances occurring over a short time (e.g., hurricanes Rita and Ike or hurricanes Laura and Delta) without separate, focused inventories. If lagged mortality from the 2011 drought occurred within the same measurement cycle, it would be grouped together with initial mortality, e.g., making it more challenging to distinguish species-specific trends. Additionally, the growth of stems that were alive at the initial measurement but died at the next measurement is not counted in growth rates here and, thus, they are likely underestimates of actual growth rate. However, if the binary process of individual stem mortality rate is adjusted to units of basal area for each stem, then that missing component of growth is accounted for rather as a transition from live to dead. Similarly, stems that were alive at the initial measurement and died before the next measurement and subsequently were removed are not counted in mortality rate calculations. Thus, mortality rates based on individual stems are likely an underestimate of actual mortality to a degree depending on the number of trees that died and were removed prior to remeasurement. Despite these limitations, long-term FIA data capture forest demographic trends well given the consistency of measurement protocols and programmatic commitments to maintain core measurements (McRoberts et al., 2005). Ultimately, long-term inventory data such as FIA are capable of revealing vulnerable species and those capable of recovery whether from short-term, acute events such as this drought (Nielsen-Gammon, 2012) or long-term, chronic dieback events (such as oak dieback in Missouri and Arkansas, e.g., Fan et al., 2008; Haavik et al., 2012).

Spatial considerations

When considering spatial scale, FIA data is capable of capturing trends from widespread disturbances and diebacks (Yang et al., 2021). Here, we can see clear evidence of the 2011 drought in east Texas, an event that spanned the entire state and was the worst ever recorded (Hoerling et al., 2013; Nielsen-Gammon, 2012). Smaller scale disturbances, i.e., on the order of a few hectares to thousands of hectares, can be more challenging to examine given the spatial scale of the FIA inventory. Since a single plot represents approximately 2400 ha of land area, any single event less than that area may not be captured on field plots. Importantly, FIA collects categorical information denoting agents of mortality and damage to individual stems and conditions surrounding plots. These codes can be used to filter certain types of disturbances or damaging agents and, when used in appropriate combinations, can answer specific forest dynamics questions (Fitts et al., 2022). Thus, trees damaged or killed by localized disturbances from pests or prescribed fire/wildfire can be filtered and examined to provide resolution on those distinct processes using data across larger spatial scales. Furthermore, considerable effort is being conducted in the field of small area estimation (SAE) to leverage large-scale inventory data such as FIA to understand forest resource changes at small scales. For example, Gaines and Affleck (2021) estimated post-wildfire tree density by borrowing FIA sample data from within the same spatial domain and across multiple measurement years, while Green et al. (2020) estimated common stand metrics in loblolly pine plantations using ancillary lidar data. Ultimately, both studies showed promise as a statistical tool requiring further development. Riley et al (2016) developed a technique using random forests to correlate forest conditions from FIA with spatial biophysical data (i.e., LANDFIRE) providing a promising means of imputing forest

conditions in unsampled areas. Notably, large-scale forest inventory data is tremendously valuable as a reference dataset when combined with airborne/spaceborne lidar, providing the means to interpolate to larger or smaller spatial scales using various modeling techniques (Babcock et al., 2016; Narine et al., 2019; Sheridan et al., 2014). Large-scale forest inventory data such as FIA provides critical estimates of forest change across large spatial scales and is increasingly becoming valuable for estimates at much smaller scales.

Logistical considerations

Given the comprehensive spatial and temporal scale of FIA, it can take one to several years for the latest annual inventory to become publicly available (Randolph et al., 2021). This makes it challenging for managers and decision-makers to conduct a rapid damage assessment of forests after a major disturbance with FIA data alone. Alternative efforts are required to capture that information within a rapid timeframe, e.g., within a few days/weeks or the growing season following the disturbance. Recent major disturbances, including the 2011 drought in Texas, Hurricane Michael in 2018 in the southeastern USA, and the 2020 derechos in Iowa, are all examples of acute disturbance events necessitating rapid assessments of forest damage which could not be met immediately via standard FIA remeasurement periods (Moore et al., 2016; Georgia Forestry Commission 2018; Florida Forest Service, 2018; Goff et al., 2021). Damage from more chronic events, such as beetle outbreaks in the western USA, impacts from invasive species, and dieback and drought events, can be monitored well using standard FIA schedules given their more gradually occurring nature (Morin et al., 2017; Shaw et al., 2005; Thompson, 2017; Vogt et al., 2016; Yang et al., 2021). However, even in the cases of acute events, long-term inventory data such as FIA still reliably provides comprehensive damage information and can be tailored to accommodate disturbance-specific inventory needs (e.g., via intensification, adapting measurements to variables of interest). Moreover, these data can be used to later validate immediate damage assessment efforts and can inform long-term trends of recovery or decline (Brandeis et al., 2022; Klockow et al., 2018).

National vs regional considerations

Although we focus on a specific region within a single state as a case study, FIA is a national program continually moving toward national consistency. Given this, it is possible to use the demographic approach we present here to provide comprehensive assessments for any number of domains, including the continental USA, states, counties, species, or ecoregions. The FIA program maintains core variables collected in a convenient hierarchical nature (i.e., plot, condition, tree) on all plots and uses rigorous detail in tracking the same plots and stems over time. Many quality control checks are in place to minimize errors and maintain long-term integrity (Pollard et al., 2006). Importantly, these repeated measurements and the hierarchical nature of FIA data conform well to hierarchical modeling approaches. For example, Salas-Eljatib and Weiskittel (2020) recently examined multiple hierarchical mortality modeling approaches to identify those that make the most effective use of data structures similar to FIA, including multiple remeasurements and inconsistent remeasurement periods. Notably, these same modeling approaches could be applied to stem-fall processes given their binary nature similar to mortality. Moreover, the long-term, consistent remeasurements provide a wealth of information that could inform growth-and-yield models, e.g., planted loblolly or naturally regenerated loblolly across its entire US range. Other studies have made important use of the FIA data structure to develop downed dead wood decomposition rates and decay class transitions across the eastern USA (Russell et al., 2013, 2014). Similar approaches can be applied to snags to better understand standing dead decomposition, fuel dynamics, and carbon estimation (Oberle et al., 2018).

Even with national consistency, there remain many regional nuances within the FIA program and data. As discussed earlier, all plots within any particular state are measured over a certain number of annual panels, 5 or 7 years in the eastern USA and 10 years in the western USA. States in the eastern USA have four or five measurements of plots whereas western states are just completing the first set of remeasurements. However, as the climate changes, the frequency of disturbances in the western USA has increased (Barrett & Long, 2021) and it is becoming increasingly important to track changes at a more frequent pace. In

part to address this, a regional effort was undertaken within FIA to collect tree cores from stems on FIA plots throughout the interior western USA (DeRose et al., 2017). This comprehensive dataset provides "annual remeasurements" of individual stems via tree ring increment measurements and extends the measurement record back decades. Current efforts are leveraging this dataset to develop models of forest response to climate change for informing future trends and management options (Heilman et al., 2022). Elsewhere, capturing forest resource changes and accounting for carbon in interior Alaska has become an increased priority for the FIA program. However, inventorying this region following core protocols is prohibitively costly and time consuming. To overcome these challenges, new approaches are being developed that combine reduced field plot sampling intensity with airborne lidar data to provide detailed, wall-to-wall inventory information at a much more favorable cost compared to standard protocols (Babcock et al., 2018). While these specific examples are not used consistently across FIA and the USA, they provide clever solutions to regional issues that could potentially be adapted elsewhere.

Conclusions

We present demographic metrics over multiple remeasurements highlighting trends, consistencies, and nuances of key forest demographic processes in east Texas. Results provide a holistic look at forest resource changes of differing species groups within the FIA inventory over a 20-year span (i.e., 2001–2021). Demographic rates of growth and mortality from large-scale disturbances, namely the 2011 drought, in Texas are captured well with FIA data providing important context to changes in forest resources. Stem-fall and removal rates also are well captured by FIA inventory protocols. However, more nuanced analyses and ancillary information are likely required to understand changes in rates and implications of these processes. Moreover, continued tracking of fallen stems remaining on plots would provide an increase and more holistic resolution to the fate of carbon. Although monitoring programs maintained by agencies, e.g., NASA, can capture and attribute large pulses of mortality to droughts or changes in forestland area, it is the specific rates of forest demographics that are truly informative to resolving important but nuanced forest resource dynamics. However, identifying diffuse lagged mortality, species- and size-specific effects, and management effects requires more detailed information as collected in field-based inventories such as FIA. By better understanding the opportunities and gaps in the framework of a field-based inventory such as FIA for tracking demographic rates, improvements in estimation and monitoring can be made which can ultimately facilitate improved forest resource management.

Acknowledgements We thank the USDA Forest Service, Northern Research Station, for providing funding for this project through a Joint Venture Agreement (# 20-JV-11242305-081) with the University of Minnesota. We also thank the folks of the USDA Forest Service, Forest Inventory and Analysis program, for their work in collecting, organizing, analyzing, and making available the data used for this paper.

Author contribution PK, CE, GD, CW, and MR contributed to research conceptualization; PK, CE, GD, CW, and MR contributed to methodology; PK contributed to data analysis and interpretation; GD and CE contributed to funding provision; PK, CE, GD, CW, and MR contributed to the writing of the original draft and review and editing.

Funding Funding for this project was provided by the USDA Forest Service, Northern Research Station, through a Joint Venture Agreement (# 20-JV-11242305-081) with the University of Minnesota.

Data availability The data used to generate the results of this work is publicly available from the US Department of Agriculture, Forest Service, Forest Inventory and Analysis (FIA) program. FIA data can be obtained via the DataMart at this link: https://apps.fs.usda.gov/fia/datamart/datamart.html.

Declarations

Competing interests The authors declare no competing interests.

Ethics approval All research pertaining to this article did not require any ethics approval or research permit(s). All authors have read and understood and have complied as applicable with the statement on "Ethical responsibilities of authors" as found in the Instructions for Authors.

Conflict of interest The authors declare no conflicts of interest.

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