



1 Species- and region-specific growth curves, forest age (2020), 2 and disturbance trends for French forests at 30 m resolution

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13 Abstract

14 Spatially explicit biomass growth curves data that jointly resolve tree species, stand age and
 15 disturbance regime are scarce, particularly at fine spatial resolutions. We present five consistent
 16 data products for metropolitan French forests around 2020, derived from a 30 m dominant-
 17 species map, the 30 m FORMS-B satellite above-ground biomass (AGB) map and the Landsat-
 18 based European Forest Disturbance Atlas: (1) 290 AGB growth curves for 28 species categories
 19 across 11 large ecological regions; (2) a 30 m forest age map for 2020 covering 17.01 Mha of
 20 forests, obtained by inverting these curves against the AGB map; (3) 3,752 harvest and natural
 21 disturbance rates and patch sizes, with their 1999–2024 trends; (4) 3,752 event-based disturbance
 22 severities (AGB loss per disturbed hectare); and (5) 1,874 rotation periods from National Forest
 23 Inventory (NFI) plot samples remeasurements. Products 3–5 cover 28 species categories across
 24 86 small ecological regions. These cover the species×region combinations present in French
 25 forests, not the full combinatorial grid. Not all 28 species categories occur in every ecological
 26 region; the totals reflect strata with forest area only. All products include uncertainties and gap-
 27 filling flags. Growth curves match National Forest Inventory-derived curves for early growth
 28 (+12±15%, 95% CI) but give a higher asymptotic biomass (+22±18%) and a later half-saturation
 29 year (+33±23%), partly reflecting differences in the representation of age extremes, as NFI
 30 measurements exclude trees <7.5 cm in diameter and we aggregate stands >60 years into a single



age class. The age map matches the NFI nationally aggregated age distribution (median offset -2 $[-20$ to $+42]$ yr). The 2020 harvested area (44 ± 6 kha) and the resulting AGB loss (2.5 ± 0.3 MtC.yr^{-1}) are of the same order as independent clear-cut estimates (62 kha from the NFI; ~ 3 MtC.yr^{-1} from satellite-mapped clear-cuts over 2018–2024). The Landsat disturbance product captures only stand-replacing and high severity events and underestimate selective logging and diffuse mortality. These products can be combined directly to model forest carbon dynamics at species and regional resolution. All data are openly available at <https://doi.org/10.5281/zenodo.23499335> (Pellissier-Tanon et al., 2026).

Keywords: forest carbon, growth curves, forest age mapping, disturbance regime, harvest, France, remote sensing, aboveground biomass

1 Introduction

French forests are among the most diverse in Europe, spanning oceanic, continental, mountain and Mediterranean climates and hosting a wide range of tree species, from fast-growing conifer plantations to old broadleaf stands (Cavaignac, 2009; IGN, 2025c). Understanding and monitoring forest carbon dynamics at fine spatial and species resolution is essential for informing climate mitigation policy, given the growing evidence that the French forest carbon sink has weakened substantially over the past decade (Ciais et al., 2026). Species composition is a key determinant of forest growth and vulnerability to disturbance (Forzieri et al., 2021; Pilli et al., 2024), yet most existing large-scale data products describing forest growth, age, and disturbance are either aggregated to broad forest types (conifer/broadleaf) or lack the spatial and species resolution needed to characterize local carbon dynamics (Besnard et al., 2021; Robinson et al., 2025).

Monitoring forest resources in France relies primarily on the National Forest Inventory (NFI), which surveys around 7,000 new semi-permanent plots annually. The NFI provides robust statistical sample-based estimates of carbon stocks, growth, and removals at national to regional scale, and sample-based statistics of species composition and stand structure. However, its sparse sampling limits reliable estimates to regional scale (around 5,000–6,000 km^2 , after pooling several annual campaigns). Without complementary remote sensing observations, NFIs therefore



59 cannot map growth, age and disturbance at finer scales, a limitation that becomes acute as
60 disturbances intensify in localized areas.

61 Several global and European forest-age products cover France, but none combines fine
62 resolution with species-specific growth. Global products range from the 0.5° Global Forest Age
63 Dataset, resolved by plant functional type (Poulter et al., 2019), to GAMI at 100 m (Besnard et
64 al., 2024) and a recently released 30 m global map separating natural and planted forests,
65 focusing only on the 0-40 years range (Wang et al., 2026). At the European scale, Vilén et al.
66 (2012) reconstructed forest age structure over 1950–2010 at 0.25° by backcasting country-level
67 age-class statistics, and Moreno et al. (2017) combined harmonised NFI data with remote
68 sensing to map mean tree age on a 0.133° grid. These products rely on globally or continentally
69 trained models that ignore species-specific growth, and most are too coarse to resolve the small,
70 fragmented stands typical of French forests. Their systematic biases when applied at national or
71 species level therefore remain largely unquantified.

72 Growth curves, which describe how stand biomass accumulates with stand age, face a similar
73 limitation. The 5 km map of (Robinson et al., 2025) is based on global upscaling of field data
74 that lack Mediterranean plots and represent natural regrowth, so it is poorly suited to France's
75 mostly managed forests. The IPCC default growth rates (IPCC, 2019) are resolved only by
76 ecozone and conifer/broadleaf type. Disturbance data are also incomplete: Landsat-based
77 archives such as the European Forest Disturbance Atlas (Viana-Soto and Senf, 2025) map when
78 and where stand-replacing disturbances occur and attribute their agents, but provide no species-
79 level or patch size information. Estimates of the biomass lost per disturbance, also called severity
80 (Kowalski et al., 2026) and of the age at harvest are likewise rarely resolved by species and
81 region, although both are needed to translate disturbed areas into carbon losses.

82 Earth observation offers a way to fill these gaps. National biomass maps are available at 30 m
83 resolution (Schwartz et al., 2023, 2026), and Landsat archives record nearly four decades of
84 stand-replacing disturbances (Viana-Soto and Senf, 2025). Combined with a dominant-species
85 map, these data resolve the fine-scale heterogeneity that NFI sampling alone cannot fully
86 resolve. Biomass maps, however, are single-year snapshots that carry no direct information on
87 stand age or growth, and they are less statistically robust than inventory-based estimates. A



88 space-for-time approach can derive growth and age from such snapshots, while NFI data provide
89 an independent, statistically robust reference against which the resulting estimates can be
90 evaluated.

91 This paper describes five data products built to fill these gaps for metropolitan France: (1) 290
92 species×region above-ground biomass (AGB) growth curves after Landsat disturbances for 28
93 tree species across 11 large ecological regions; (2) a 30 m forest age map for 2020 obtained by
94 inverting these curves against observed AGB; (3) 3,752 trend-based harvest and natural
95 disturbance rates and patch sizes for 28 species across 86 small ecological regions over 1999–
96 2024; (4) 3,752 harvest and natural disturbance severities (AGB loss per disturbed hectare) for
97 the same species and regions around 2020; and (5) 1,874 species- and region-specific rotation
98 periods derived from National Forest Inventory (NFI) remeasurements. Remote sensing products
99 1-4 were validated against independent estimates, when possible. All products are harmonised:
100 they share the same 28 species, AGB definition, and 2020 reference year, and use a nested
101 regionalisation of 86 small ecological regions grouped into 11 larger ones. They can therefore be
102 combined directly to support forest carbon modelling.

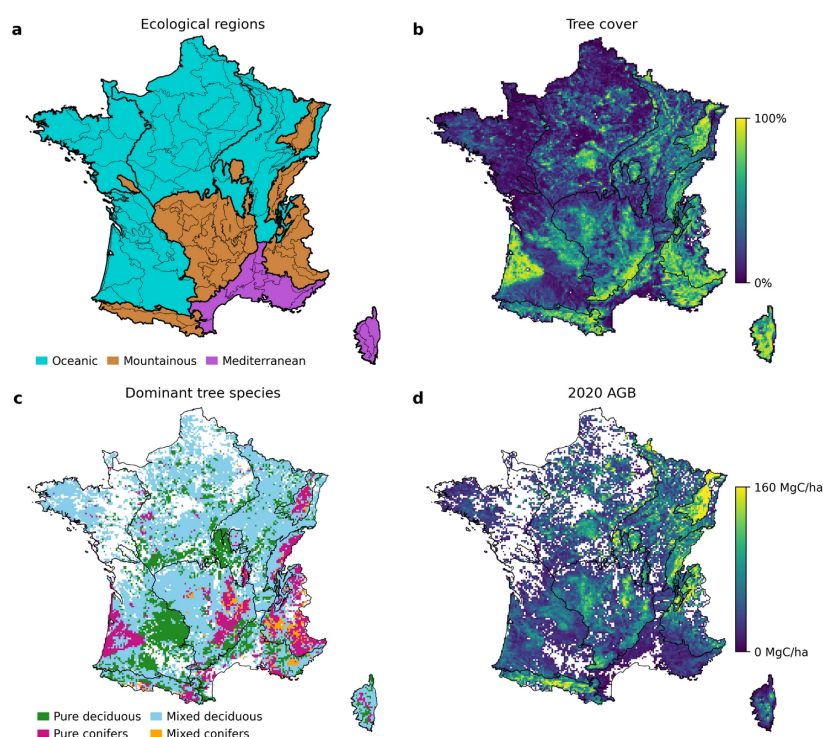
103 **2 Data and methods**

104 **2.1 Study area**

105 France's 17 Mha of forest (IGN, 2025c) can be divided into 86 small ecological regions (SERs),
106 ecologically coherent units defined by climate, geology, and topography, aggregated into 11
107 large ecological regions (GRECOs) (Cavaignac, 2009) (Fig. 1). Growth was quantified by
108 species at the GRECO scale, the finest that provides enough stands of all ages for the space-for-
109 time approach. Disturbances were quantified at the SER scale, the finest that still captures
110 enough disturbance events for a robust signal.. French forests are highly diverse in climate
111 (Atlantic, Mountainous, Mediterranean), management intensity (from intensive silviculture to
112 unmanaged stands), and ownership (three-quarters privately owned) (IGN, 2025c). Mixed
113 deciduous stands (8.2 Mha) dominate, with deciduous oaks (2.7 Mha) as the single largest
114 species group, alongside fir/spruce (0.9 Mha), maritime pine (0.8 Mha), beech (0.7 Mha), Scotch
115 pine (0.6 Mha), evergreen oak (0.5 Mha), chestnut (0.3 Mha), and Douglas fir (0.3 Mha) (IGN,
116 2018). The NFI estimates total carbon stock at 1.26 GtC for living trees and 0.11 GtC for dead



117 wood (IGN, 2025c). Only ~60% of production forest is harvestable with standard machinery,
 118 while ~32% lies on steep, distant, or weakly bearing terrain (IGN, 2025c), so a substantial share
 119 of standing biomass is only marginally accessible.



120

121 **Figure 1.** (a) Nested 86 small ecological regions inside 11 large ecological regions. Maps of tree cover (b),
 122 dominant tree species (c), AGB in 2020 (d), aggregated at 5km. Pixels with a forest cover smaller than 10%
 123 were excluded from (c) and (d). Map data / credits: ecological regions from IGN SER/GRECO; tree species
 124 and forest mask from BD Forêt® v2 © IGN (IGN, 2018); AGB from FORMS-B (Schwartz et al., 2023).

125 2.2 Input datasets

126 Five datasets underpin the data products presented here.

127 2.2.1 Dominant species category map

128 The BD Forêt v2 dataset (IGN, 2018) was used to define the dominant tree species category and
 129 as a forest mask. The shapefile was rasterized onto the common 30 m grid used throughout. This
 130 national map provides 30 m explicit information on dominant species or genus, leaf type, and



131 tree-cover class, derived from manual photo-interpretation of aerial imagery acquired between
132 2007 and 2018. Early-stage or recently established stands may be underrepresented or
133 misclassified. Two BD Forêt classes describing recently cut forest without tree cover were
134 excluded from the species-resolved products (growth curves and disturbance metrics) but
135 retained in the age map after reassignment. They cover 0.63 Mha (3.4% of the BD Forêt forest
136 area 18.26 Mha). The 28 species categories are described in Table 1. The resulting mask covers
137 17.35 Mha.

Forest type	Species categories	Area (Mha, %)
Pure deciduous	Deciduous oak, beech, evergreen oak, chestnut, poplar, other deciduous, black locust	4.54 (26.2%)
Mixed deciduous	Mixed deciduous, mainly deciduous, mixed deciduous in island, open deciduous, open mixed	8.14 (46.9%)
Pure conifers	Fir/spruce, maritime pine, Scotch pine, Douglas fir, Austrian pine, Aleppo pine, larch, mountain/stone pines, other non-pine, other pines	3.12 (18.0%)
Mixed conifers	Mainly conifers, mixed conifers, open conifers, mixed non-pine, mixed pines, mixed conifers in island	1.55 (8.9%)

138 **Table 1. Aggregation of the 28 BD Forêt species categories into the four forest types used in this study. Full**
139 **definitions in Table S1. Fir/spruce and mountain/stone pines are each a single class.**

140 2.2.2 Above-ground biomass map

141 The FORMS-B AGB map (Schwartz et al., 2023) was used to construct growth curves and to
142 invert the age map. The FORMS dataset provides an above-ground biomass (FORMS-B) map
143 for France for 2020, derived from a deep-learning model combining GEDI lidar with Sentinel-
144 1/2 imagery. Biomass was estimated using two NFI-calibrated allometric equations (deciduous
145 vs. conifer). The 30 m resolution was selected to capture the fragmented spatial distribution of
146 forest species, and a France-specific product was preferred over continental or global biomass
147 maps for its higher accuracy.



148 **2.2.3 Historical disturbance maps**

149 Young forest ages (age ≤ 35 years) were derived from the European Forest Disturbance Atlas
 150 (EFDA) v3.0.0 (Viana-Soto and Senf, 2025), which provides annual Landsat-based maps of
 151 forest disturbances across Europe (1985–2024). EFDA distinguishes wind/bark-beetle, fire, and
 152 harvest disturbance events using a classification-based approach applied to harmonized summer
 153 Landsat composites, with an overall F1-score of 0.89 and $<1\%$ error for undisturbed areas.
 154 Commission and omission errors for disturbed areas are 17.3% and 22.5% respectively,
 155 improving to 10.6% after 2000.

156 **2.2.4 Global forest age map**

157 For forests 35 years and older, the GAMI v3.1 dataset was used, providing global forest-age
 158 distributions for 2020 at 100 m resolution (Besnard et al., 2024). Age estimates were generated
 159 using XGBoost machine-learning models trained on more than 40,000 forest-inventory plots
 160 worldwide, together with biomass and height data, Landsat-based disturbance history, remote-
 161 sensing inputs, and climate variables. To account for aleatoric and epistemic uncertainty,
 162 multiple models with varied hyperparameters were used, and twenty realizations of biomass
 163 inputs with controlled perturbations were applied to simulate natural variability; we used the
 164 ensemble mean and standard deviation as the age estimate and its associated uncertainty.

165 **2.2.5 National Forest Inventory age and biomass measurements**

166 IGN annually monitors around 7,000 newly selected semi-permanent forest plots, each 30 m in
 167 diameter, systematically distributed across mainland France as part of the NFI, with each plot
 168 revisited five years after its first survey (IGN, 2025a). For a representative subset of trees, tree-
 169 core samples provide direct age measurements for dominant individuals; where a tree is too
 170 small to core and has only recently become measurable, its age is derived from a count of
 171 growth-ring vertices. These tree-level measurements were used to estimate species-specific
 172 rotation periods (Section 2.5.3). Separately, NFI-based growth curves were built as a validation
 173 dataset from sampling-based NFI statistics. AGB-vs-age curves were extracted either per tree
 174 species at the national scale, or per large ecological region pooling all species, from first-visit
 175 age and AGB measurements using the same space-for-time approach and Chapman-Richards



176 fitting scheme as the remote-sensing curves (Section 2.3), with uncertainty propagated from the
 177 NFI sampling design.

178 **2.3 Growth curves of biomass vs. age**

179 Growth curves relating AGB to stand age since the last Landsat-based disturbance were built per
 180 species categories $\times$ large ecological region (28 species categories $\times$ 11 regions; (Cavaignac,
 181 2009)) using a space-for-time approach (Pellissier-Tanon et al., 2024) pairing AGB densities
 182 over 30 m pixels that experienced a past disturbance, for a given species category and region.
 183 This approach substitutes chronosequences for direct long-term monitoring: pixels of the same
 184 species and region but different ages are treated as snapshots along a common recovery
 185 trajectory, allowing the age-AGB relationship to be reconstructed from a single-year AGB map
 186 rather than requiring repeated measurement of the same stand over decades.

187 For ages below or equal to 35 years, disturbance-derived ages were used with 1-year-wide bins,
 188 defined as the number of years since the most recent detected disturbance, assuming immediate
 189 regrowth:

$$190 \quad \text{age} = 2020 - (\text{year of last disturbance}) \quad (1)$$

191 with this definition applied for ages not exceeding 35 years, i.e. within the reliable temporal
 192 depth of the Landsat disturbance record used here. For ages of more than 35 years, GAMI ages
 193 were used, after masking all disturbed pixels. For each species/region pair, AGB percentiles were
 194 computed within each age bin. Young EFDA ages remain affected by disturbance-year
 195 uncertainty and by overestimation of AGB in low-biomass stands in the 2020 map; we therefore
 196 discarded ages < 2 and subtracted a partial offset of $0.85 \times \text{AGB}(t=2)$ from later EFDA AGB
 197 percentiles, as a correction for early-map bias. For each per species $\times$ region the 25th, 50th and
 198 75th AGB percentiles were fitted jointly with a Chapman-Richards growth model:

$$199 \quad \text{AGB}_\tau(t) = \text{AGB}_{\max}(\tau) (1 - e^{-bt})^c \quad (2)$$

200 sharing shape parameters b and c governing early growth rate and curvature across percentiles,
 201 with the asymptotic maximum biomass $\text{AGB}_{\max}$ constrained to increase with the percentiles τ
 202 ((Robinson et al., 2025; Sáez-Cano et al., 2021; Zeng et al., 2026). For ages above 90 years, any



subsequent decline of more than 5% below the running peak AGB was treated as a decreasing tail and removed before fitting; younger ages were left unchanged. Three summary metrics were derived from each fitted curve for validation and comparison purposes: the asymptotic maximum biomass (AGB_{max} , from Eq. 2 directly), the time to reach half of AGB_{max} (t_{50} , obtained by inverting Eq. 2 at $AGB = 0.5 \times AGB_{max}$), and the mean growth rate over the first 30 years of the curve ($R_{30} = AGB(30)/30$). Local fits were retained only when $\min(R_{25}^2, R_{50}^2, R_{75}^2) > 0.2$. Species categories $\times$ regions pairs failing this criterion, or lacking a fit, were treated as missing. Missing curves with positive forest area were gap-filled in two steps: first by copying the kept fit from the largest-area donor of the same species category within the same bioclimatic group (Oceanic, Mountainous or Mediterranean, Fig. 2); if none was available, from a fixed climate neighbour (Mediterranean J/K from F; Oceanic B/F from G). 50 missing species/region combinations were filled from neighbouring regions, corresponding to 3.4% of the forest area. The 50th-percentile fitted curve was retained as the central estimate, with the 25th- and 75th-percentile curves defining the lower and upper bounds of the uncertainty range. Growth-curve metrics are reported as the value from the 50th-percentile curve, followed in brackets by the values from the 25th- and 75th-percentile curves.

2.4 Forest age map

The 100-m GAMI age map underestimates the share of young stands: relative to the NFI distribution, both its lower quartile and its median are shifted markedly higher (+86% and +47%, respectively), and it covers substantially less forest area (6.6 Mha) than our species-based forest mask (17.35 Mha). We therefore constructed an improved 30-m age map by inverting the 50th-percentile Chapman-Richards growth curve of each species category $\times$ large ecological region against the observed 2020 AGB map, i.e. solving Eq. 2 for t given an observed AGB value and the species $\times$ region's fitted (AGB_{max} , b , c):

$$age = \frac{-1}{b} \log \left[1 - \left(\frac{AGB}{AGB_{max}} \right)^{\frac{1}{c}} \right] \quad (3)$$

Where the observed AGB exceeded the value predicted by the growth curve at its asymptote, stand age was capped at 99.9% of the curve's maximum AGB (i.e. the value of age at which the



230 curve reaches $0.999 \times AGB_{max}$), to avoid the inversion diverging for pixels with AGB approaching
 231 or exceeding AGB_{max} .

232 To avoid discontinuities at large-ecological-region boundaries, where two neighbouring regions
 233 can have markedly different growth-curve parameters for the same species category, which
 234 would otherwise produce a sharp jump in estimated age at the boundary, a smooth blending
 235 scheme was applied. For each GRECO, a soft membership weight was derived from the signed
 236 distance to the region boundary (tanh-scaled over a 30 km range zone), approaching 1 deep
 237 inside the region, 0.5 on the boundary, and 0 beyond 30 km outside. Age was computed for
 238 every 30m pixel under the growth-curve parameters of each region in turn, and the final pixel-
 239 level age was taken as the weighted mean of these region-specific age estimates, using the
 240 blending weights described above. This ensures a smooth transition in the age map across large-
 241 ecological-region boundaries rather than an abrupt one. Before age estimation, pixels lacking
 242 species information were reassigned: newly cut closed forest to mixed deciduous, and newly cut
 243 open forest to open deciduous forest. Pixels with age = 0 and zero aboveground biomass were set
 244 to no-data unless associated with a disturbance in 2019–2021, to remove spurious 1.25 Mha of
 245 zero-age estimates while retaining likely true young regenerating stands. The resulting forest
 246 area is 17.01 Mha.

247 Lower and upper age uncertainty bounds were obtained by repeating the inversion (Eq. 3) using
 248 the 25th- and 75th-percentile growth curves respectively, in place of the 50th-percentile curve,
 249 propagating the growth-curve uncertainty range into the age map. Age statistics are reported as
 250 the value from the p50 map, followed in brackets by the same statistic computed on the p25 and
 251 p75 maps.

252 **2.5 Trends of harvest and natural disturbances**

253 Harvest and natural disturbance (fire, wind, biotic attacks) rates, patch sizes, and severities were
 254 quantified per species categories and small ecological regions. Rates and patch sizes come from
 255 the EFDA archive over 1999–2024, and severities from the 2020 AGB map. Losses were
 256 decomposed into two independent components: (i) a trend-based estimate of the disturbed area
 257 and patch size, with its linear 1999–2024 trend, and (ii) a constant severity (AGB loss per
 258 disturbed hectare) estimated with a space-for-time approach (Kowalski et al., 2026).



259 2.5.1 Disturbance rate and patch-size

260 Annual disturbance area and mean patch size were extracted from EFDA for each species pair
 261 (category × small ecological region), separating harvest from natural disturbance (fire,
 262 wind/biotic) using the EFDA agent classification. The disturbance rate is the disturbed area
 263 expressed as a percentage of the pair's forest area. Patches are groups of 8-connected disturbed
 264 pixels of the same year and agent. The analysis starts in 1999 because EFDA commission and
 265 omission errors drop markedly after 2000 (Viana-Soto and Senf, 2025).

266 Trend significance was tested with the Mann-Kendall test ($p < 0.05$). Where significant, the time
 267 series was fitted with the Sen slope (the median of all pairwise slopes) and the Kendall-Theil
 268 intercept. Where not significant, the series was represented by its median, with a slope of zero.
 269 The trend-based 2020 estimate is the fitted value at 2020, or the median for non-significant
 270 series. Trends are reported relative to this 2020 value.

271 Uncertainty in the fitted slope and intercept was estimated from the mean squared error of the
 272 residuals. For median-represented series, the standard error of the median was:

$$273 \quad SE = 1.253 \sigma / \sqrt{n} \quad (4)$$

274 where σ is the sample standard deviation across years and n is the number of years with
 275 disturbance observations. A finite-sample correction factor was applied to this standard error,
 276 equal to 1 for $n \geq 20$, $\sqrt{(n-1)/(n-3)}$ for $3 < n < 20$, and 2 otherwise. The ± 1 year uncertainty in
 277 disturbance dating was not propagated into the trends, as Sen's slope over 25 years is robust to
 278 such small shifts.

279 Mann-Kendall and Sen's slope were computed when at least 4 years with disturbance
 280 observations were available ($p < 0.05$). Series with fewer than 10 observation years retained the
 281 median level for 2020 but were assigned a null slope so short series do not contribute a fitted
 282 trend. When no disturbance was recorded over 1999–2024, the rate was set to 0. Regional,
 283 forest-type and national values are forest-area-weighted means of the species category × small
 284 ecological region estimates. The share of forest area with a significant trend is the summed forest
 285 area of pairs with a significant Mann-Kendall test, divided by the total forest area.



286 2.5.2 Disturbance severity and AGB loss

287 Absolute and relative severity, defined as the AGB loss per disturbed hectare, one year after
 288 disturbance, were estimated following Kowalski et al. (2026) from the 2020 AGB map and
 289 EFDA, separately for each disturbance agent and species category $\times$ small ecological region pair.
 290 Pixels disturbed after 2020 (2021–2024) were used to characterize pre-disturbance AGB density,
 291 and pixels disturbed before 2020 (2016–2019) were used to characterize post-disturbance AGB
 292 density. Absolute severity was defined as:

$$293 \quad S = q_{50}^{before} - 0.5(q_{50}^{after} + q_{05}^{after}) \quad (5)$$

294 and relative loss as:

$$295 \quad S_{rel} = S / q_{50}^{before} \quad (6)$$

296 where q_{50}^{before} is the 50th percentile of pre-disturbance AGB (pixels with AGB = 0 are excluded
 297 from the pre-disturbance cohort), and q_{50}^{after} and q_{05}^{after} are the 50th and 5th percentiles of post-
 298 disturbance AGB. Sampling uncertainties assumed asymptotic normality of sample quantiles.

299 Let σ_{before} and σ_{after} be the sample standard deviations of AGBD in the pre- and post-disturbance
 300 cohorts, and n_{before} and n_{after} the corresponding disturbed-pixel counts, treated as independent. The
 301 standard error of S combined the variance of the pre-disturbance median with the variance of
 302 $S = q_{50}^{before} - 0.5(q_{50}^{after} + q_{05}^{after})$ is

$$303 \quad SE(S) = \sqrt{\frac{\pi}{2} \frac{\sigma_{before}^2}{n_{before}} c(n_{before})^2 + 1.81 \frac{\sigma_{after}^2}{n_{after}} c(n_{after})^2} \quad (7)$$

304 where the factor 1.81 is the normal asymptotic variance of $0.5(q_{50}^{after} + q_{05}^{after})$, and $c(n) = 1$ if $n \geq 20$,
 305 $\sqrt{(n-1)/(n-3)}$ if $3 < n < 20$, and 2 if $n \leq 3$. The standard error of relative severity was obtained by:

$$306 \quad SE(S_{rel}) = \frac{1}{q_{50}^{before}} \sqrt{SE(S)^2 - Var(q_{50}^{before}) \left(2 \frac{S}{q_{50}^{before}} - \left(\frac{S}{q_{50}^{before}} \right)^2 \right)} \quad (8)$$

307 with $Var(q_{50}^{before}) = \frac{\pi}{2} \frac{\sigma_{before}^2}{n_{before}} c(n_{before})^2$.



Species category $\times$ region $\times$ agent trios with no matched pre-/post-disturbance pair were gap-filled first with the same species $\times$ agent estimate at the regional scale, then with the national species $\times$ agent estimate. Gap filling affected 850 species $\times$ region $\times$ agent trios corresponding to 1.9% of the total forest area. National and forest-type severities are forest-area-weighted means.

AGB-loss intensity was computed per species category $\times$ small ecological region pair and agent as the trend-based 2020 disturbance rate $\times$ absolute severity. Total AGB loss is intensity $\times$ forest area. Regional, forest-type and national totals are sums over pairs. Uncertainties in AGB loss were propagated using first-order (Taylor series) error propagation from two independent sources: disturbance-rate uncertainty and severity uncertainty.

2.5.3 Rotation period

Species category- and small-ecological-region-specific rotation periods were estimated from 2005–2024 NFI remeasurements. Trees recorded as harvested between two successive plot visits were identified. Their removal age was approximated as the plot's dominant-tree cored age at the first visit plus 2.5 years (the approximate mid-point between visits). BD Forêt categories were matched to NFI species based on the plot composition description. After filtering for at least 5 trees per species category $\times$ small ecological region, matching was possible for 21 of the 28 categories, yielding 450 values. Eight BD Forêt categories had no direct NFI harvest-age sample and were assigned the nearest mapped category: mixed deciduous in island, mainly deciduous and open deciduous from mixed deciduous; mountain/stone pines, mixed conifers in island, mainly conifers and open conifers from mixed conifers; and open mixed from open deciduous, itself taken from mixed deciduous. For the other species categories, missing pairs were gap-filled from large-ecological-region, then national, estimates. Gap filling affected 1,425 rotation periods (32.7% of forest area): 950 by large-region or national cascade (15.7% of forest area; 106 from unmatched species, 2.9%; 844 from matched species missing regional data, 12.8%), and 475 by species-alias assignment at the regional level (17.0% of forest area). Mean harvest age and its standard deviation (σ) were computed per species category $\times$ small ecological region from the n harvested trees. The uncertainty of the rotation period is the standard error of the mean, $SE = \sigma / \sqrt{n}$, with the same finite-sample correction as in Eq. 4. The files also provide σ , which describes the spread of harvest ages.



337 **2.5.4 Uncertainty propagation and aggregation**

338 For all disturbance quantities (rates, trends, patch sizes, severities and AGB losses), aggregation
339 to regional, forest-type or national values used the same rules: area-weighted means (or sums for
340 total AGB loss). Row-level standard errors were combined assuming full correlation among
341 region×species category pairs that share the same species category (species block) and
342 independence across species. For disturbance and rotation products, figures show ± 1 SE and
343 values in the text are 95 % confidence intervals ($\pm 1.96 \times \text{SE}$).

344 **2.6 Variance partitioning**

345 To assess whether growth and disturbance metrics are structured more by species category or by
346 region, we computed one-way η^2 (between-group sum of squares / total sum of squares) for each
347 metric and each grouping factor (species category, ecological region and, for disturbances,
348 agent), treating each fitted value equally (species $\times$ large ecological region for growth curves;
349 species $\times$ small ecological region $\times$ agent for disturbances). 95% confidence intervals were
350 obtained by Monte Carlo perturbation of each value within its standard error for disturbance
351 metrics and 25th–75th percentile range for growth metrics ($n = 1000$ draws), and significance by
352 label-permutation tests ($n = 1000$ permutations). Gap-filled values were excluded from the
353 variance partitioning, and rotation periods (harvest only) were excluded from the agent analysis.

354 **3 Data records**

355 The five products are distributed as one 30 m raster and four tables (Table 2). The raster is a
356 GeoTIFF in [Lambert-93, EPSG:2154], aligned with the FORMS-B grid, with NoData =
357 [−9999]. Tables are CSV files keyed by species category code (Table S1), large ecological
358 region (GRECO) code and small ecological region (SER) code, following IGN nomenclature.
359 Uncertainties are given as 1 SE for disturbance and rotation tables; p25 and p75 curves and maps
360 for growth and age. The column gap_filled flags gap-filled values (0 = original, 1 = regional, 2 =
361 national, 3 = species-alias assignment, rotation only).

Product	Format	Spatial unit	Key variables (units)	Records	Gap-filled (records, % forest)
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					area)
Growth curves	CSV	Species × large region	AGB_{max} (MgC.ha ⁻¹), b, c, t_{50} (yr), R_{30} (MgC.ha ⁻¹ .yr ⁻¹)	290	50; 3.4%
Age map	GeoTIFF, 30 m	30 m pixel	age_p50, age_p25, age_p75 (yr)	17.01 Mha	Inherited from growth curves
Disturbance rate and patch size	CSV	Species × small region × agent	rate_2020 (%.yr ⁻¹), slope (%.yr ⁻¹), patch_2020 (ha), patch_slope (%.yr ⁻¹), MK p-value, SEs	3,752	0; 0%
Severity	CSV	Species × small region × agent	S (MgC.ha ⁻¹), S_rel (%), SEs, n_before, n_after	3,752	850; 1.9%
Rotation period	CSV	Species × small region	mean harvest age (yr), SE (yr), n_trees	1,874	1,425; 32.7%

Table 2. Data files. Products are linked by species category code (Table S1), large and small ecological region codes (IGN), and disturbance agents. Uncertainties are given as 1 SE for disturbance and rotation tables, and as 25th- and 75th-percentile curves and maps for growth and age. The files include a gap-filling flag. The full column dictionary is in the repository README.

4 Data description and validation

4.1 Forest growth curves

We derived 290 (3.4% of forest area gap-filled) growth curves across 11 large ecological regions and 28 tree species categories. Fig. 2 illustrates these curves for the two most widespread pure species, deciduous oak and fir/spruce (Fig. 2 a-b). Growth is jointly shaped by regional environmental conditions and species identity (Fig. 2 d-f). Panels g–i summarize the fitted curves across all species categories and regions using three metrics: asymptotic maximum biomass (AGB_{max}), time to half-asymptote (t_{50}), and mean growth rate over the first 30 years (R_{30}).

The area-weighted national mean is 112 ± 12 MgC.ha⁻¹ (95% CI) for AGB_{max} , 1.2 ± 0.1 MgC.ha⁻¹.yr⁻¹ for R_{30} , and 57 ± 7 years for t_{50} . AGB_{max} is on average higher for pure and mixed



deciduous stands than for pure and mixed conifers (Fig. 2g), though with substantial variability across forest types (from evergreen oaks at $34 \pm 24 \text{ MgC.ha}^{-1}$ to Douglas fir at $168 \pm 20 \text{ MgC.ha}^{-1}$; Fig. S1). A similar pattern holds for t_{50} , which is longer for deciduous and shorter for conifer stands, ranging from poplars (16 ± 8 years) to other non-pine conifers (105 ± 8 years) (Fig. 2i). R_{30} depends less on forest type, spanning $0.2 \pm 0.1 \text{ MgC.ha}^{-1}.\text{yr}^{-1}$ (evergreen oaks) to $2.5 \pm 0.5 \text{ MgC.ha}^{-1}.\text{yr}^{-1}$ (poplars) (Fig. 2h).

Our remote-sensing curves agree well with NFI-derived curves, whether matched by species at the national scale ($AGB_{max} +22 \pm 18\%$, $R_{30} +12 \pm 15\%$, $t_{50} +33 \pm 23\%$, relative difference to NFI averaged over the species; 95% CI) or by ecoregion pooling all species ($AGB_{max} +31 \pm 21\%$, $R_{30} -2 \pm 11\%$, $t_{50} +63 \pm 22\%$, averaged over the ecoregions). Note that the NFI-derived curves are subject to some limitations at the youngest and oldest ages: trees with a diameter below 7.5 cm are not measured, while stands older than 60 years are grouped into a single age class. Compared with Robinson et al. (2025), our AGB_{max} estimates are similar ($+8 \pm 22\%$ vs. ours, $+32 \pm 19\%$ vs. NFI), but their curves largely overestimate R_{30} ($+174 \pm 100\%$ vs. ours, $+147 \pm 62\%$ vs. NFI) and underestimate t_{50} ($-67 \pm 4\%$ vs. ours, $-47 \pm 7\%$ vs. NFI) (Fig. S2).

Across France, variance in growth-curve metrics is more strongly structured by species category identity than by ecological region, suggesting that intrinsic species traits play a larger role in shaping growth trajectories than regional climate and soil conditions. Using one-way η^2 (Sect. 2.6), species identity explains more variance on average than ecoregion (mean $\eta^2 = 34 \pm 7\%$ vs. $27 \pm 14\%$, 95% CI). Relative weights vary by metric: AGB_{max} shows moderate species dominance (species $\eta^2 = 38 \pm 4\%$ vs. region $23 \pm 3\%$). R_{30} remains similarly high for both factors ($38 \pm 4\%$ vs. $41 \pm 3\%$), and t_{50} shows the sharpest contrast favouring species effect (27% vs. 16%). Both species and regional effects are significant for all three metrics (permutation tests, $p = 0.001$). Altogether the age at half asymptote is especially species-driven and less shaped by regional conditions than early growth rate or long-term asymptotic biomass.

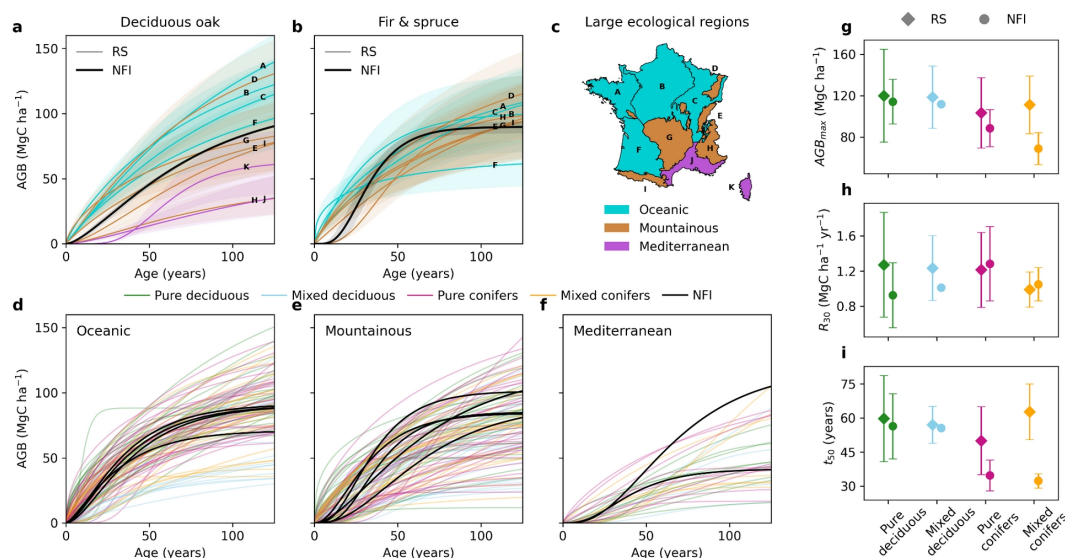


Figure 2. Tree species and regional AGB growth. AGB growth: regional fits from remote sensing and country level data for NFI for deciduous oak (a) and fir/spruce (not distinguished in the species map) (b). (c) Large ecological regions. Remote sensing speciescategory×regional-level and NFI regional-level fitted curves for oceanic (d), mountainous (e), and mediterranean (f) regions. (g-i) Comparison of fitted growth curves metrics by forest type between remote-sensing species category×regional-level (diamonds) and species-level NFI (circles) growth curves for maximum AGB (AGB_{max} ; g), early growth rate (R_{30} ; h), and time to reach 50% of AGB_{max} (t_{50} ; i). For each forest type, metrics are area-weighted mean across regions and species categories; bars show the area-weighted SD across the species×regions. AGB is expressed per hectare of forested area.

4.2 Forest age map

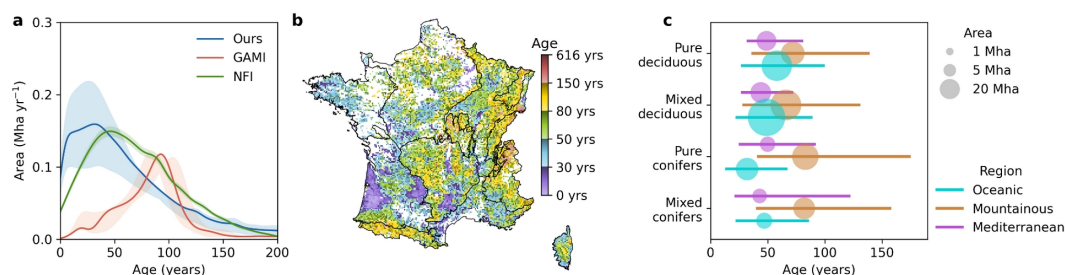
Our 30 m species category- dependent age map covers substantially more forest area (17.0 Mha) than GAMI (6.6 Mha), and corrects GAMI's systematic aging of young-stand ages relative to NFI (Q1 +30 [+17 to +43] years, median +28 [+16 to +40] years, Q3 +13 [+1 to +26] years), attributable to GAMI's coarser (100 m) resolution and globally-trained, non-species-specific calibration. Bracketed ranges are uncertainty from lower and upper age maps (and, where relevant, GAMI standard deviation over the members or NFI area-sampling error).

At the country scale, our map is similar to the NFI across the distribution (difference in age quantiles, map – NFI: Q1 –8 [–16 to +8] yr, median –2 [–20 to +42] yr, Q3 +20 [–20 to +175] yr), better than GAMI (Fig. 3a, S3c). Stratified by species or by region, agreement holds at Q1 and the median but weakens at Q3: Stratified by species, the NFI-area-weighted mean offset of



our map relative to NFI is modest at Q1 (−7 [−13 to −2] yr) and near zero at the median (+7 [−6 to +21] yr) but positive at Q3 (+44 [−3 to +92] yr), while GAMI shows a large, near-constant positive offset (Q1 +33 [+30 to +35] yr, median +33 [+31 to +36] yr, Q3 +32 [+30 to +35] yr). Stratified by region, the offset is small at Q1 (−5 [−10 to −1] yr) and the median (+3 [−9 to +16] yr) but positive at Q3 (+39 [−19 to +98] yr), whereas GAMI retains a large positive bias throughout (Q1 +31 [+27 to +34] yr, median +27 [+23 to +32] yr, Q3 +28 [+24 to +32] yr) (Figs. S4–5).

According to our 30 m age map, French forests have a median age of 56 [40–99] years, with pronounced spatial heterogeneity (IQR 80 [48–220] years) driven by management intensity and species composition (Figs. 3bc, S6). Pure conifers in Oceanic forests show the lowest ages (Q1–Q3: 14 [12–19]–66 [46–145] years), driven by intensively managed regions such as maritime pine plantations in les Landes. Mediterranean forests are similarly young (Q1–Q3: 29 [22–43]–79 [51–207] years), reflecting frequent disturbance-driven age reset despite low management intensity. Mountainous forests are the oldest (Q1–Q3: 35 [24–61]–144 [82–538] years), consistent with limited management at high elevation. Nationally, conifers have similar lower quartiles to deciduous stands (Q1 = 28 [21–43] vs. 27 [20–40] yr) but extend to older ages (Q3 = 126 [75–296] vs. 101 [66–255] yr), reflecting both short-rotation plantations and old mountain stands. Pure and mixed stands have similar age distributions (Q1 = 29 [21–44] vs. 26 [19–39] yr; Q3 = 112 [70–263] vs. 103 [66–261] yr). Landscape-scale age is also highly heterogeneous: the mean spatial coefficient of variation within 5 km cells is 105 [93–105]%, with an IQR of 84 [67–84]–122 [113–122]% (Fig. S3). The largest values occur in Les Landes and the Massif Central, and the smallest in the continental East. Substantial variability in age structure among individual tree species categories, within and across the broad forest-type and regional groupings, further confirms that species identity, not just region or forest type, is a necessary axis for characterizing France's forest age structure.



447

448 **Figure 3. Forest age map. (a) Age distributions from remote sensing, NFI, and GAMI. Forest area:**
 449 **Ours RS: 17.01 Mha, GAMI: 6.6 Mha, NFI: 15.6 Mha. Note that GAMI's smaller mapped area**
 450 **reflects the 100 m forest mask, and NFI covers production forest only. (b) Remote-sensing forest**
 451 **age map of France, aggregated at 5 km, and pixels with a forest cover smaller than 10% were**
 452 **excluded. (c) Forest age by dominant composition class (pure deciduous, pure conifers, mixed**
 453 **deciduous, mixed conifers) and ecological region (Oceanic, Mountainous, Mediterranean).**
 454 **Horizontal bars are the age-distribution [Q1–Q3] within each composition×region class, bubble position**
 455 **indicates median age and bubble size is proportional to forest area. Colors denote ecological**
 456 **regions.**

457 4.3 Disturbance metrics

458 4.3.1 Harvest and natural disturbance rate and patch size

459 We estimated harvest and natural disturbance (i.e. fire, storms/biotic attacks) rates and patch
 460 sizes for 28 species categories across 86 small ecological regions. Rates are defined as the area
 461 disturbed per year as a percentage of forest area and patch size the mean area of contiguous
 462 disturbed pixels. We computed the recent dynamics of their trends over the last 25 years (1999–
 463 2024). This yields 3,752 species category × region × agent estimates (Fig. 4, S7–S8). Trends
 464 were tested with the Mann-Kendall test and fitted with Sen's slope. All 2020 values reported
 465 below are trend-based estimates, not the disturbance observed in that year. They therefore reflect
 466 long-term signals and smooth out episodic peaks such as the 2018-2022 bark-beetle outbreaks or
 467 the 2022 Landes fires. Disturbance rates are expressed as the area disturbed in 2020 as a
 468 percentage of forest area. Values in the text are given as $\pm 95\%$ CI, and figures show ± 1 SE.

469 The trend-based estimate of harvested area in 2020 is 44 ± 6 kha.yr⁻¹, or $0.26 \pm 0.03\%$ of total
 470 forest area, somewhat consistent with the independent NFI-reported harvest rate of 62 kha.yr⁻¹
 471 for stands with >90% canopy loss (IGN, 2025b). Harvest rates are highest in les Landes
 472 ($1.0 \pm 0.3\%$) and lower in central and southern France ($0.04 \pm 0.01\%$), and vary by forest type:
 473 highest for pure conifers ($0.5 \pm 0.1\%$), intermediate for mixed conifers ($0.35 \pm 0.08\%$), and lowest



474 and similar for pure and mixed deciduous forests ($0.20 \pm 0.05\%$ and $0.18 \pm 0.05\%$) (Fig. 4ab). At
 475 the species category level, rates range from $0.06 \pm 0.05\%$ for mountain pines and $0.09 \pm 0.06\%$ for
 476 beech to $0.9 \pm 0.3\%$ for maritime pine and $0.7 \pm 0.2\%$ for poplars, with intermediate values for
 477 deciduous oak ($0.16 \pm 0.08\%$), chestnut ($0.5 \pm 0.1\%$), fir/spruce ($0.5 \pm 0.2\%$) and Douglas fir
 478 ($0.6 \pm 0.2\%$); the full species category ranking is given in Fig. S8a.

479 Nationally, the harvest rate shows a significant increase of $1.6 \pm 0.8 \text{ \%} \cdot \text{yr}^{-1}$ relative to its 2020
 480 value. A significant increasing trend is detected over 37% of the forest area, a significant
 481 decrease over only 3%, and no significant trend over the remaining 59%. The increase is
 482 regionally heterogeneous, ranging from $4.1 \pm 0.8 \text{ \%} \cdot \text{yr}^{-1}$ in Massif Central to a marginally
 483 significant decrease in Corsica ($-2.0 \pm 1.7 \text{ \%} \cdot \text{yr}^{-1}$). By forest type, it is significant for pure and
 484 mixed conifers (2.5 ± 0.7 and $2.4 \pm 1.1 \text{ \%} \cdot \text{yr}^{-1}$), while pure and mixed deciduous forests (0.8 ± 1.5
 485 and $0.9 \pm 2.0 \text{ \%} \cdot \text{yr}^{-1}$) show no significant national trend (Fig. 4d,e). At the species level, the
 486 strongest increase is in fir/spruce ($3.6 \pm 2.2 \text{ \%} \cdot \text{yr}^{-1}$) and mixed conifers ($3.3 \pm 2.0 \text{ \%} \cdot \text{yr}^{-1}$), with a
 487 significant increase also in mixed conifers non pine ($3.3 \pm 2.0 \text{ \%} \cdot \text{yr}^{-1}$), Aleppo pine (2.9 ± 1.1
 488 $\text{ \%} \cdot \text{yr}^{-1}$), Scotch pine ($2.9 \pm 1.9 \text{ \%} \cdot \text{yr}^{-1}$) and Douglas fir ($2.5 \pm 1.6 \text{ \%} \cdot \text{yr}^{-1}$). (Fig. S8b).

489 The trend-based estimate of natural disturbance in 2020 is $0.20 \pm 0.08\%$ of forest area, about
 490 three-quarters of the harvest, and is spatially concentrated in les Landes ($2.4 \pm 0.8\%$) with
 491 scattered impact elsewhere (Fig. 4c). Pure conifers are by far the most affected forest type
 492 ($0.7 \pm 0.3\%$), followed by mixed conifers ($0.2 \pm 0.2\%$), then pure and mixed deciduous forests
 493 ($0.09 \pm 0.10\%$ and $0.07 \pm 0.10\%$) (Fig. 4b). Only four species categories show a significant national
 494 natural disturbance rate: maritime pine ($2.1 \pm 1.0\%$), poplar ($0.3 \pm 0.2\%$), chestnut ($0.20 \pm 0.15\%$)
 495 and mountain pines ($0.07 \pm 0.07\%$) (Fig. S8a).

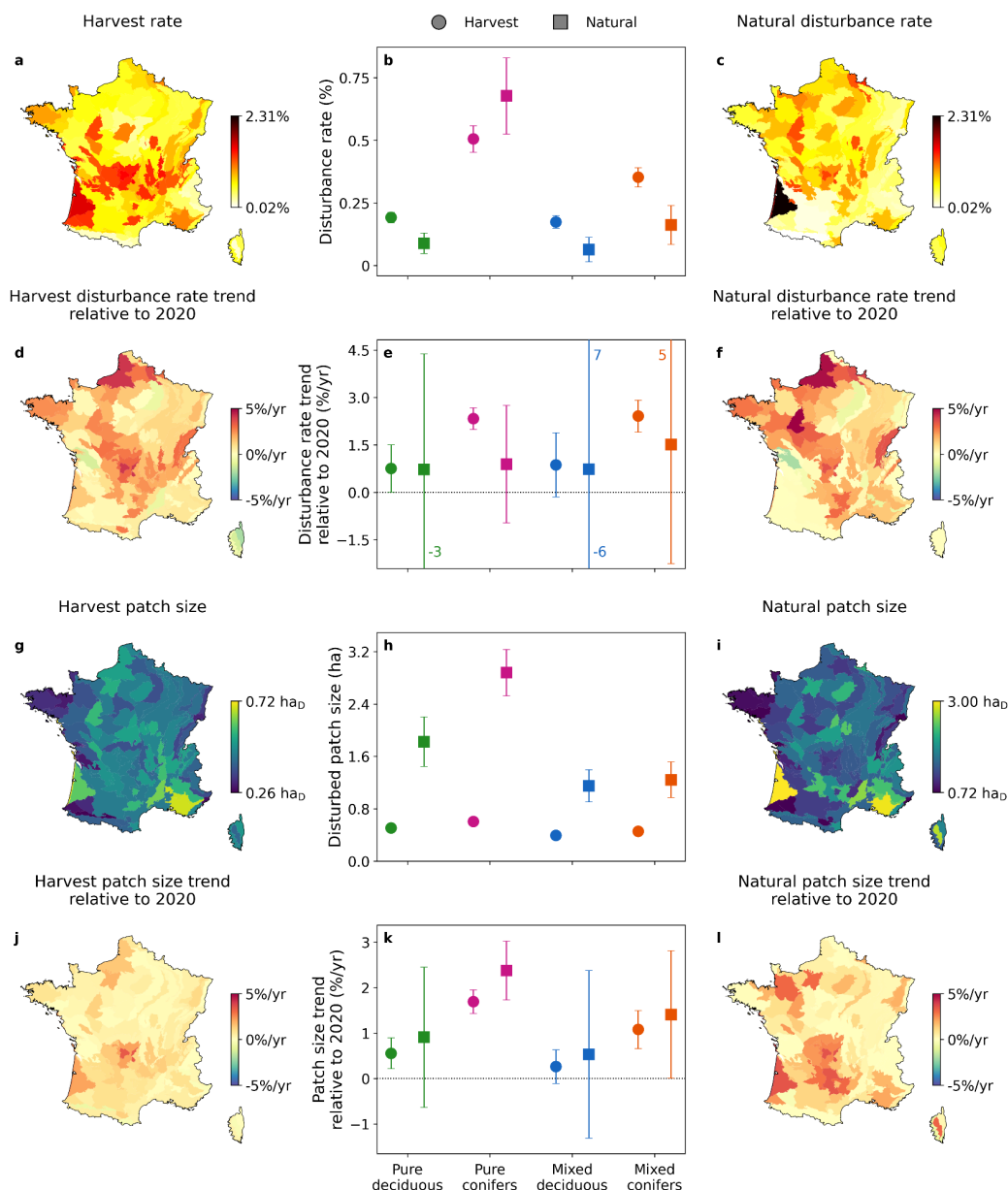
496 Unlike harvest, natural disturbance shows no significant national trend relative to its 2020 rate
 497 ($1 \pm 3 \text{ \%} \cdot \text{yr}^{-1}$). A significant increase is detected locally, over 23% of forest area, a significant
 498 decrease over 3%, and no significant trend over the remaining 75% (Fig. 4f). As with harvest,
 499 conifers show larger (but still non-significant at national scale) point estimates than deciduous
 500 forests (mixed conifers: $2 \pm 8 \text{ \%} \cdot \text{yr}^{-1}$, significant over 20% of their area; pure conifers: $1 \pm 4 \text{ \%} \cdot \text{yr}^{-1}$,
 501 38% of their area) compared to pure and mixed deciduous forests ($1 \pm 11 \text{ \%} \cdot \text{yr}^{-1}$, 19%; 1 ± 13
 502 $\text{ \%} \cdot \text{yr}^{-1}$, 22%) (Fig. 4e). At the species category level, the most spatially extensive increases are in



503 Aleppo pine ($5 \pm 4 \text{ \%} \cdot \text{yr}^{-1}$, significant over 87% of its area; also the only significant national
 504 mean), fir/spruce ($4 \pm 12 \text{ \%} \cdot \text{yr}^{-1}$, 50%) and Douglas fir ($2 \pm 15 \text{ \%} \cdot \text{yr}^{-1}$, 32%) (Fig. S8b).

505 Harvest patches are significantly smaller than natural disturbance patches nationally (0.46 ± 0.03
 506 ha vs. 1.6 ± 0.3 ha) and range from 0.27 ± 0.05 ha (Provence) to 0.98 ± 0.30 ha (les Landes) (Fig.
 507 4g,i). Harvest patch size is larger for pure conifers (0.60 ± 0.08 ha) than pure deciduous
 508 (0.50 ± 0.05 ha) or mixed stands (deciduous: 0.39 ± 0.04 ha; conifers: 0.45 ± 0.07 ha) (Fig. 4h), and
 509 ranges from 0.17 ± 0.04 ha (mixed deciduous in island) to 0.7 ± 0.2 ha (maritime pine) (Fig. S8c).
 510 Harvest patches expand nationally by $0.7 \pm 0.4 \text{ \%} \cdot \text{yr}^{-1}$ of their 2020 size. This increase is driven
 511 by pure conifers ($1.6 \pm 0.6 \text{ \%} \cdot \text{yr}^{-1}$), with a significant contribution from mixed conifers (1.0 ± 0.9
 512 $\text{ \%} \cdot \text{yr}^{-1}$) and no significant trend in pure or mixed deciduous forests (0.5 ± 0.7 and $0.2 \pm 0.8 \text{ \%} \cdot \text{yr}^{-1}$)
 513 (Fig. 4j,k). At the species category level, harvest patch size increases significantly for fir/spruce
 514 ($1.8 \pm 0.9 \text{ \%} \cdot \text{yr}^{-1}$), maritime pine ($1.9 \pm 1.2 \text{ \%} \cdot \text{yr}^{-1}$), Scotch pine ($1.9 \pm 1.5 \text{ \%} \cdot \text{yr}^{-1}$) and Aleppo pine
 515 ($2.2 \pm 0.8 \text{ \%} \cdot \text{yr}^{-1}$) (Fig. S8d).

516 Natural disturbance patch size ranges from 0.61 ± 0.37 ha (Atlantic coast) to 5.7 ± 3.5 ha in fire-
 517 prone les Landes. In pure conifers, natural disturbance patches (2.8 ± 0.7 ha) are about five times
 518 larger than harvest patches (0.60 ± 0.08 ha). In deciduous and mixed stands the difference is
 519 smaller (pure deciduous: 1.8 ± 0.7 vs. 0.50 ± 0.05 ha; mixed deciduous: 1.1 ± 0.5 vs. 0.39 ± 0.04 ha;
 520 mixed conifers: 1.3 ± 0.5 vs. 0.45 ± 0.07 ha) (Fig. 4h). At the species category level, the largest
 521 natural-disturbance patches occur in Aleppo pine (4.2 ± 2.2 ha), followed by maritime pine
 522 (3.8 ± 1.9 ha), Austrian pine (3.1 ± 2.1 ha), larch (2.6 ± 1.7 ha) and Scots pine (2.2 ± 1.7 ha) (Fig.
 523 S8c). Natural disturbances patch size shows no significant national trend ($1.2 \pm 1.7 \text{ \%} \cdot \text{yr}^{-1}$ of 2020
 524 size) but increases significantly in pure conifers ($2.5 \pm 1.4 \text{ \%} \cdot \text{yr}^{-1}$); at the species category level,
 525 only maritime pine ($3.3 \pm 2.1 \text{ \%} \cdot \text{yr}^{-1}$) and Aleppo pine ($3.5 \pm 2.3 \text{ \%} \cdot \text{yr}^{-1}$) show a significant increase
 526 (Fig. S8d).



527

528 **Figure 4. Harvest and natural disturbance rate and patch size. (a–c) Disturbance rates in 2020, read from the**
 529 **fitted 1999–2024 linear trend, for harvest (a) and natural disturbances (c), and by forest type (b), as a**
 530 **percentage of the corresponding forest area. (d–f) Slope of the 1999–2024 trend in disturbance rate for**
 531 **harvest (d) and natural disturbances (f), and by forest type (e), as a percentage of the 2020 rate per year. (g–i)**
 532 **Mean patch size in 2020, read from the fitted 1999–2024 trend, for harvest (g) and natural disturbances (i),**
 533 **and by forest type (h). (j–l) Slope of the 1999–2024 trend in patch size for harvest (j) and natural disturbances**



(l), and by forest type (k), as a percentage of the 2020 value per year. Maps show area-weighted mean values per small ecological region, aggregated from the underlying species category-level estimates; bar charts show national area-weighted means per forest type, shown separately for harvest and natural disturbances. Error bars denote ± 1 SE. ha_D denotes disturbed forest area. Corresponding uncertainty maps are shown in Fig. S7.

4.3.2 Harvest and natural disturbance severity

Disturbance severity, defined as the AGB lost per disturbed unit area (pre- minus post-disturbance AGB), was estimated with the space-for-time approach of Kowalski et al. (2026). It was computed for 28 species categories across 86 small ecological regions, yielding 3,752 species category $\times$ region $\times$ agent estimates (1.9% of forest area gap-filled). Severity is reported in absolute terms ($MgC.ha^{-1}$ of disturbed area) and relative to pre-disturbance AGB (Fig. 5a–f, S9–S10).

Harvest severity averages 50 ± 1 $MgC.ha^{-1}$ when weighted by forest area, with a relative severity of $79 \pm 1\%$ of pre-disturbance AGB. Nationally both follow the spatial pattern of standing AGB, from large values in the north-east (99 ± 2 $MgC.ha^{-1}$, $89 \pm 2\%$) to small values in the Mediterranean (6.6 ± 0.2 $MgC.ha^{-1}$, $60 \pm 2\%$) (Fig. 5a,d). By forest type, harvest absolute severity is similar across classes: mixed deciduous (47 ± 1 $MgC.ha^{-1}$, $78 \pm 2\%$), pure deciduous (50 ± 2 $MgC.ha^{-1}$, $77 \pm 2\%$), mixed conifers (48 ± 2 $MgC.ha^{-1}$, $80 \pm 4\%$) and pure conifers (57 ± 1 $MgC.ha^{-1}$, $84 \pm 1\%$) (Fig. 5b,e).

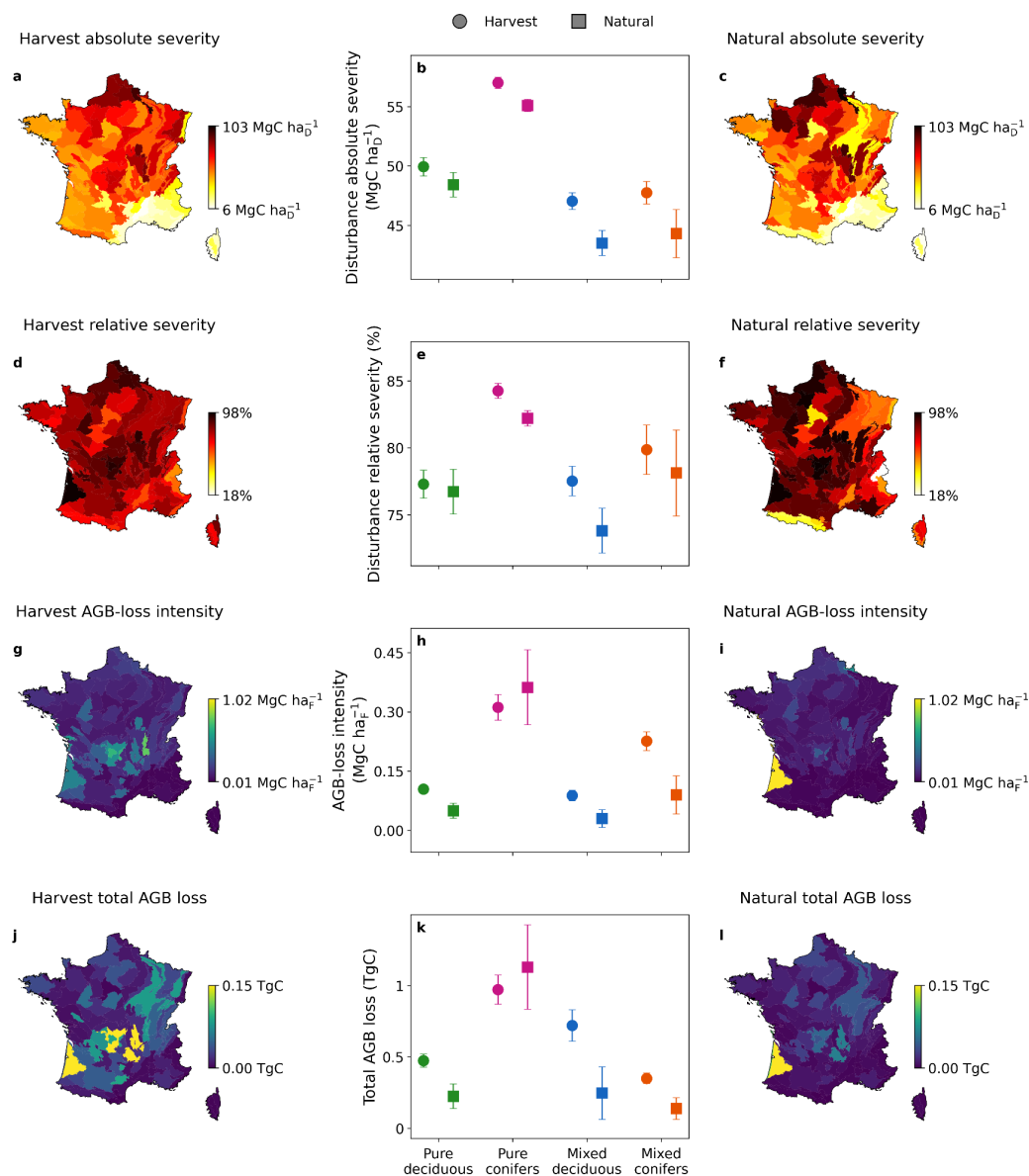
Nationally, natural severity is 47 ± 1 $MgC.ha^{-1}$ ($77 \pm 2\%$), with a somewhat similar spatial pattern to harvest, from the north-east (104 ± 2 $MgC.ha^{-1}$, $95 \pm 2\%$) to the Mediterranean (5.7 ± 0.3 $MgC.ha^{-1}$, $53 \pm 3\%$) (Fig. 5c,f). For natural disturbances, absolute severity is highest for pure conifers (55 ± 1 $MgC.ha^{-1}$, $83 \pm 1\%$), then pure deciduous (49 ± 2 $MgC.ha^{-1}$, $77 \pm 3\%$), mixed conifers (45 ± 4 $MgC.ha^{-1}$, $79 \pm 6\%$) and mixed deciduous (44 ± 2 $MgC.ha^{-1}$, $74 \pm 3\%$), all similar to the corresponding harvest values (Fig. 5b,e).

At the species category level, harvest and natural absolute severity remain close for most types (typically within ≤ 7 $MgC.ha^{-1}$). Absolute severity spans a wide range, from poplar (harvest 103 ± 4 $MgC.ha^{-1}$; natural 101 ± 7 $MgC.ha^{-1}$), Douglas fir (102 ± 4 ; 106 ± 4), mixed non pine (83 ± 5 ; 84 ± 5) and fir/spruce (82 ± 2 ; 76 ± 2) down to Aleppo pine (21 ± 1 ; 21 ± 1), mountain pines (19 ± 6 ; 0 ± 0) and evergreen oak (6 ± 1 ; 6 ± 4) (Fig. S10). Relative severity is highest for poplar ($98 \pm 2\%$; $97 \pm 4\%$), maritime pine ($97 \pm 1\%$; $96 \pm 1\%$) and Douglas fir ($94 \pm 3\%$; $94 \pm 3\%$), and lowest for larch



564 (52±5%; 56±3%), evergreen oak (58±6%; 53±18%) and mountain pines (62±14%; 2±0%) (Fig.
565 S10a,b).

566 Compared with the independent Planet-based severity maps of Kowalski et al. (2026) over
567 France, our rate-weighted absolute severity averages 49 MgC.ha⁻¹ versus 31 MgC.ha⁻¹ in
568 Kowalski et al., and relative severity 78% versus 55% of pre-disturbance AGB (forest-area-
569 weighted SER means) (Fig S11). Despite this systematic positive offset (+18 MgC.ha⁻¹), the
570 spatial patterns agree closely ($r = 0.88$ absolute; $r = 0.68$ relative).



571

572 **Figure 5. Harvest and natural disturbance severity and AGB lost.** (a-c) Absolute disturbance severity (AGB
 573 pre disturbance - AGB post disturbance) per disturbed forest area for harvests (a) and natural disturbances
 574 (c), and by forest type (b). (d-f) Relative disturbance severity for harvest (d) and natural disturbances (f), and
 575 by forest type (e), expressed as a percentage of the pre disturbance AGB. (g-i) AGB loss per forest area
 576 (trend-based estimate of disturbance rate in 2020 × disturbance severity) for harvest (g) and natural
 577 disturbances (i), and by forest type (h). (j-l) AGB loss (trend-based estimate of disturbance rate in 2020 ×
 578 disturbance severity × forest area) for harvest (j) and natural disturbances (k), and by forest type (l). Maps



show area-weighted mean values per small ecological region, aggregated from the underlying species category-level estimates; bar charts show national area-weighted means per forest type, shown separately for harvest and natural disturbances. Error bars denote ± 1 SE. ha_F denotes total forest area and ha_D disturbed forest area. Corresponding uncertainty maps are shown in Fig. S9.

4.3.3 Harvest and natural disturbance carbon losses

We derived two complementary metrics from the 2020 trend-based disturbance rate (Sect. 4.3.1) and severity (Sect. 4.3.2), both computed at species category $\times$ region level. AGB-loss intensity (rate $\times$ severity) measures how much AGB each hectare of forest loses per year, independent of forest extent (Fig. 5g–i). Total AGB loss (intensity $\times$ forest area) measures where losses matter in absolute terms and is summed to regional and national totals (Fig. 5j–l). A small forest type can therefore dominate intensity but not totals, and vice versa. National values are sums over species category $\times$ region pairs, not products of national means.

Using the 2020 harvest rate from the fitted 1999–2024 trend, national harvest represents an AGB-loss intensity of 0.15 ± 0.02 MgC.ha⁻¹ for a total AGB loss of 2.5 ± 0.3 MtC, of the same order as the ~ 3 MtC.yr⁻¹ estimated from satellite-mapped clear-cuts over 2018–2024 (Canopée, 2026). Harvest AGB-loss intensity ranges from the north-eastern Massif Central (0.79 ± 0.13 MgC.ha⁻¹) to Corsica (0.01 ± 0.00 MgC.ha⁻¹) (Fig. 5g). By forest type, pure conifers show the largest harvest loss nationally (0.98 ± 0.21 MtC) despite occupying only 3.1 Mha of forest (18%) due to their high AGB-loss intensity (0.31 ± 0.07 MgC.ha⁻¹); mixed deciduous forests, though dominant in area (8.1 Mha, 47%), contribute 0.73 ± 0.23 MtC (0.09 ± 0.03 MgC.ha⁻¹); pure deciduous forests contribute 0.48 ± 0.11 MtC (4.5 Mha, 26%, 0.11 ± 0.02 MgC.ha⁻¹), and mixed conifers the least (0.35 ± 0.08 MtC, 1.5 Mha, 9%, 0.23 ± 0.05 MgC.ha⁻¹) (Fig. 5h,k). At the species category level, AGB-loss intensity is highest for mixed non pine (0.81 ± 0.31 MgC.ha⁻¹), poplar (0.71 ± 0.22 MgC.ha⁻¹), Douglas fir (0.61 ± 0.23 MgC.ha⁻¹), maritime pine (0.45 ± 0.17 MgC.ha⁻¹), fir/spruce (0.40 ± 0.16 MgC.ha⁻¹) and chestnut (0.26 ± 0.07 MgC.ha⁻¹) (Fig. S10c). In total, however, the largest losses come from widespread categories: mixed deciduous (0.54 ± 0.21 MtC), fir/spruce (0.36 ± 0.15 MtC) and maritime pine (0.36 ± 0.13 MtC) (Fig. S10d).

Using the 2020 natural disturbance rate from the fitted 1999–2024 trend, natural disturbances cause an AGB loss of 1.8 ± 0.7 MtC, about 70% that of harvest. Losses are concentrated in les Landes (1.05 ± 0.37 MgC.ha⁻¹; 0.65 ± 0.23 MtC) and near zero in the Pyrenees (0.01 ± 0.01

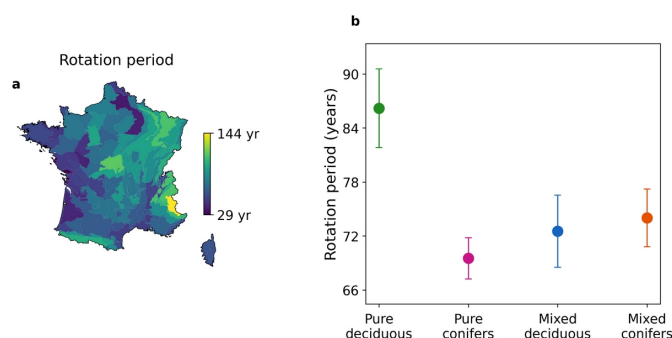


MgC.ha⁻¹) (Fig. 5i,l). Pure conifers account for most of the national loss (1.18 ± 0.60 MtC; 0.38 ± 0.19 MgC.ha⁻¹), though with wide uncertainty. Mixed deciduous (0.25 ± 0.38 MtC; 0.03 ± 0.05 MgC.ha⁻¹), pure deciduous (0.22 ± 0.17 MtC; 0.05 ± 0.04 MgC.ha⁻¹) and mixed conifers (0.15 ± 0.15 MtC; 0.10 ± 0.10 MgC.ha⁻¹) contribute little (Fig. 5h,k). At the species category level, maritime pine has the highest and only well-constrained intensity and total (1.02 ± 0.52 MgC.ha⁻¹; 0.77 ± 0.40 MtC). Douglas fir (0.38 ± 0.70 MgC.ha⁻¹) and fir/spruce (0.22 ± 0.44 MgC.ha⁻¹; 0.20 ± 0.39 MtC) have high but poorly constrained estimates (Fig. S10c,d).

4.3.4 Rotation period

Rotation periods, the mean age at which trees are harvested, were estimated from 2005–2024 NFI remeasurements for 21 species categories across 86 small ecological regions (section 2.5.3), yielding 1,874 species category $\times$ region values (32.7% of forest area gap-filled).

The rotation period averages 76 ± 4 years nationally (95% CI), larger in mountainous areas (Alps, 98 ± 6 years) and smaller in Les Landes (42 ± 2 years) (Fig. 6a). Rotations tend to be longer for pure deciduous (86 ± 9 years) than for mixed deciduous (73 ± 8 years), pure conifers (70 ± 4 years), and mixed conifers (74 ± 6 years) (Fig. 6b). By species category, rotation periods range from poplars (30 ± 4 years) to larches (151 ± 67 years), beeches (110 ± 17 years), and deciduous oaks (96 ± 14 years) (Fig. S13).



626

627 **Figure 6. Rotation period. (a,b) Rotation period for harvests (a) and by forest type (b). Maps show area-**
 628 **weighted mean values per small ecological region, aggregated from the underlying species-level estimates; bar**
 629 **charts show national area-weighted means per forest type. Error bars denote ± 1 SE. Corresponding**
 630 **uncertainty map is shown in Fig. S12.**

4.3.5 Variance partitioning



Variance in disturbance metrics is only weakly structured on average by species category, small ecological region or agent, but the dominant factor depends strongly on the metric. Using the same one-way η^2 (Sect. 2.6; equal weight per species $\times$ SER $\times$ agent cell; gap-filled values excluded), averaged over disturbance rate, patch size, severity and AGB-loss intensity (and their trends), region explains $10 \pm 1\%$ of the variance, species category $11 \pm 1\%$, and agent $3 \pm 1\%$. Absolute severity is strongly structured by species and region ($31 \pm 1\%$ and $29 \pm 1\%$, both $p < 0.001$) with only a weak agent effect ($1 \pm 0\%$, $p < 0.001$). Relative severity is region-led ($13 \pm 1\%$ vs. $9 \pm 1\%$ species, both $p < 0.001$; agent $4 \pm 0\%$, $p < 0.001$). Disturbance rate shows small but significant species and region effects ($6 \pm 1\%$, $p < 0.001$; $3 \pm 1\%$, $p = 0.016$), with no clear agent effect ($0.1 \pm 0.1\%$, $p = 0.119$). Patch size is led by agent ($15 \pm 2\%$, $p < 0.001$) rather than species or region ($7 \pm 1\%$ and $3 \pm 1\%$), reflecting larger patches for natural disturbances. AGB-loss intensity is similarly explained by species and region ($12 \pm 5\%$ and $11 \pm 2\%$, both $p < 0.001$; agent $p = 0.063$). Rotation period (harvest only; local NFI matches) is jointly structured by species and region ($48 \pm 4\%$ and $47 \pm 5\%$, both $p < 0.001$). Temporal trends remain weakly structured: for the rate trend only species is significant ($2 \pm 0\%$, $p = 0.036$; region $p = 0.310$; agent $p = 0.946$); for the patch-size trend both species and region are significant ($8 \pm 1\%$ and $5 \pm 1\%$, both $p < 0.001$; agent $p = 0.580$). Unlike growth, absolute disturbance severity carries a clear dual species–region signature, while agent identity matters mainly for patch size.

5 Data quality, uncertainties and usage guidance

5.1 Growth curves

Our median growth curves agree reasonably with NFI-derived curves for early growth rate ($R_{30} +12 \pm 15\%$, 95% CI), but show higher AGB_{max} ($+22 \pm 18\%$) and later t_{50} ($+33 \pm 23\%$). Part of this offset may be attributed to limitations in the representation of the youngest and oldest age classes in the NFI-derived curves, as trees with diameters below 7.5 cm are excluded and stands older than 60 years are grouped into a single age class. The curves of Robinson et al. (2025) give similar AGB_{max} as ours ($+8 \pm 22$), but underestimates t_{50} ($-67 \pm 4\%$) and strongly overestimate R_{30} ($+174 \pm 100\%$), consistent with their calibration on naturally regenerating, mostly tropical forests with no French or mediterranean plots (Fig. S2).



Four error sources affect the curves. First, beyond 35 years, stand age comes from GAMI, so GAMI age errors propagate into the old part of each curve. Second, radar/optical saturation at high biomass (Joshi et al., 2017; Mutanga et al., 2023) limits AGB precision in mature stands. Third, the AGB map uses generic allometric equations (Schwartz et al., 2023), which likely compresses true between-species differences, particularly in AGB_{max} . Species effects reported here are therefore conservative. Fourth, space-for-time sampling relies on stand-replacing disturbances. It therefore overrepresents intensively managed, fast-rotation species, at the expense of less intensively managed and less common species, whose curves rest on fewer young-stand pixels. Background mortality and selective logging (e.g. thinning) are not detected at 30 m, but they are implicitly embedded in the curves. Stand age is defined as time since the last stand-replacing disturbance, so the AGB observed at a given age is already net of all mortality and partial harvests since that event. The curves therefore describe net AGB accumulation under current management and mortality regimes, not potential growth. The NFI curves share this property, since they also rely on observed standing AGB. Users combining these curves with separate mortality or thinning terms should avoid double counting.

5.2 Age map

Our 30 m map resolves 2.6 times more forest area than GAMI (17.0 vs. 6.6 Mha). The gap comes mainly from small and fragmented stands that GAMI's 100 m forest mask leaves out. GAMI also overestimates ages relative to the NFI(Q1 +30 [+17 to +43] yr, median +28 [+16 to +40] yr, Q3 +13 [+1 to +26] yr). This bias is consistent with its coarser resolution and its globally trained, non-species-specific calibration. It persists across species (Q1 +33 [+30 to +35] yr, median +33 [+31 to +36] yr, Q3 +32 [+30 to +35] yr) and regions (Q1 +31 [+27 to +34] yr, median +27 [+23 to +32] yr, Q3 +28 [+24 to +32] yr). It is most pronounced in mountain regions and for less common species such as Douglas fir and fir/spruce, which global training data represent poorly (Figs. S4–5).

Our map removes this bias at the low and central parts of the distribution. Stratified by species, offsets are −7 [−13 to −2] yr at Q1 and +7 [−6 to +21] yr at the median. Stratified by region, they are −5 [−10 to −1] yr at Q1 and +3 [−9 to +16] yr at the median. Agreement weakens at Q3, where both stratifications show a positive offset (species +44 [−3 to +92] yr; region +39 [−19 to



+98] yr). Three factors likely contribute. First, the adopted NFI's open-ended ≥ 60 -year bin limits how precisely Q3 can be estimated in the first place. Second, ages are capped at $0.999 \times AGB_{max}$, which truncates the age of mature stands in above average fertility environments. Third, the AGB map itself is biased. Generic deciduous/conifer allometries miss species-specific wood density and architecture, and the optical and radar signals saturate in dense canopies. Because growth curves flatten near AGB_{max} , even small AGB errors there translate into large age errors. Age definitions also differ between the two datasets. NFI age refers to the dominant cohort of each plot, and young trees are not sampled. Our age is derived from pixel-averaged AGB, which mixes canopy and understorey and can shift the apparent age. The main limitation is circularity: ages come from inverting curves fitted to the same AGB map. In addition, curve segments beyond 35 years rely on GAMI ages, so any residual GAMI bias propagates into old-stand estimates. Ages of old, high-biomass stands should therefore be read as indicative, and users should rely on the provided uncertainty bounds rather than on the central estimate alone.

5.3 Harvest and natural disturbances

Trend reliability is limited by EFDA detection errors: 17.3% commission and 10.6% omission after 2000. Slow-acting disturbances such as bark-beetle mortality may be detected one or more years after the attack, shifting their timing. Agent attribution is also imperfect: where salvage logging follows windthrow or bark-beetle outbreaks, the same area can be classified as harvest or natural disturbance. This also inflates recent harvest trends in affected regions.

National harvest area (44 ± 6 kha.yr⁻¹) and AGB loss (2.5 ± 0.3 MtC) are somewhat consistent with independent estimates for clear-cuts (Canopée, 2026; IGN, 2025b). This agreement reflects the fact that EFDA predominantly captures stand-replacing events at 30 m resolution, so harvest trends describe clear-cuts rather than the full range of management practices (Shchur et al., 2017; Viana-Soto and Senf, 2025). Comparison with 1.5 m FORMSpOT-Δ disturbance polygons quantifies what is missed: disturbances below 0.1 ha (about 1 Landsat pixel) account for 97% of disturbance events and 39% of disturbed area in France (2015–2023), so Landsat-based products such as EFDA miss more than a third of the disturbed area (Schwartz et al., 2026). These undetected events include selective logging, thinning, small-scale mortality and diffuse drought-induced dieback (e.g. in chestnut, beech or fir). The omission is concentrated in mountain



718 regions (Alps, Pyrenees, Massif Central), where sub-0.1 ha events dominate, whereas clear-cut-
 719 dominated regions such as the Landes are well captured. As a result, our disturbance rates are
 720 underestimated and patch sizes are biased toward larger events. These datasets should therefore
 721 be interpreted as describing stand-replacing or near-stand-replacing events only, and total
 722 disturbance-related AGB loss is likely underestimated, particularly where selective silviculture
 723 dominates. Finally, severity is estimated by space-for-time as a constant AGB loss per disturbed
 724 hectare for each species $\times$ small ecological regions, which does not capture the temporal or
 725 spatial variation in disturbance intensity between events.

726 Our severities are systematically higher than those of Kowalski et al. (2026) (+18 MgC.ha⁻¹; +23
 727 percentage points; Sect. 4.3.2). This offset likely reflects (i) different input AGB maps, (ii) our
 728 use of medians instead of means for pre- and post-disturbance AGB, which reduces sensitivity to
 729 extreme values, and (iii) stratification by species category, which pairs pre- and post-disturbance
 730 pixels within homogeneous stands instead of pooling all forests within 10 km cells. Since both
 731 products share the same spatial gradient, our severities are robust in relative terms but should be
 732 used with caution as absolute values, particularly when combined with disturbance areas that
 733 include partial disturbances.

734 Unlike harvest, natural disturbance rates could not be validated against an independent national
 735 dataset. The NFI does not report a disturbed-area statistic comparable to EFDA's stand-replacing
 736 events, and agent-specific inventories are partial (fire) or report volumes rather than area (bark
 737 beetles, storms). These estimates should therefore be considered unvalidated, reflected by their
 738 large uncertainties.

739 Linear trends over 25 years are sensitive to extreme years. The series starts with the December
 740 1999 Lothar and Martin storms and ends with the 2018–2022 drought and bark-beetle crisis.
 741 Because disturbances are episodic, linear fits capture average tendencies but not pulse events.
 742 Harvest trends reflect past economic and policy conditions (wood prices, bioenergy demand,
 743 forest policy) that are not represented explicitly. Rotation periods derived from NFI
 744 remeasurements reflect past practices and may shift with climate adaptation, for example toward
 745 shorter rotations or species replacement.



746 Extrapolating these disturbance rate trends assumes stationarity in disturbance risk, which likely
747 underestimates future natural disturbances: climate change is expected to increase their
748 occurrence non-linearly, particularly for wildfire (Grünig et al., 2026; Patacca et al., 2026).
749 Interactions between disturbances, such as drought triggering bark-beetle outbreaks and
750 increasing fire risk, are not represented.

751 **5.4 General limitations**

752 All products depend on the national species category map. Misclassification of the dominant
753 species category propagates into every species category-resolved product. The input datasets also
754 cover different periods (AGB for 2020, species category map for 2007-2018, disturbances up to
755 2024). Stands disturbed after species category mapping may have been replanted or regenerated
756 with a different species category, so their species category label may not match the stand
757 observed in the 2020 AGB map. This mainly affects young stands in intensively managed, fast-
758 rotation regions. The products describe forests in their current 2020 state: forest area and species
759 category composition are held fixed, and land-use change, species turnover and CO₂ fertilization
760 are not represented. Finally, forest management is not explicitly represented. Management
761 effects are captured only implicitly, through the species category- and region-specific structure
762 of the growth, age and disturbance data. Users requiring explicit management regimes or
763 dynamic species composition should combine these products with dedicated scenario models.

764 **6 Conclusion**

765 These data products provide, for the first time at national scale, a consistent species category-
766 and region-resolved description of how French forests grow, how old they are, and how they are
767 disturbed. Because growth, age, and disturbance are derived from the same AGB, species
768 category and disturbance maps, they can be combined directly without harmonization, which
769 makes them well suited to spatially explicit carbon budgeting. Two findings matter most for
770 users. First, growth is structured more by species than by region, so aggregating by forest type or
771 ecological region alone risks blending distinct growth strategies. Second, disturbance severity
772 (AGB loss per disturbed hectare) increases with pre-disturbance biomass and varies strongly
773 between regions, so carbon losses cannot be inferred from disturbed area alone. Three limitations



774 remain, each with a clear path forward. The disturbance datasets capture stand-replacing events
775 only; integrating very high-resolution products such as FORMSpOT- Δ would extend them to
776 thinning, selective logging and diffuse mortality. Species-specific allometry in the AGB map
777 would sharpen between-species contrasts in growth. An updated species map would reduce
778 mismatches in recently disturbed stands. Because the framework relies on openly available
779 satellite and inventory data, it can be updated as new AGB and disturbance maps become
780 available, and transferred to other European countries with comparable inventories.

781 **Data availability**

782 All data products are openly available on Zenodo at <https://doi.org/10.5281/zenodo.23499335>
783 (Pellissier-Tanon et al., 2026). Input datasets are available from their providers: BD Forêt v2
784 from <https://geoservices.ign.fr/bdforet> (IGN, 2018), FORMS from
785 <https://doi.org/10.5281/zenodo.7840108> (Schwartz et al., 2023), EFDA from
786 <https://doi.org/10.5281/zenodo.13333034> (Viana-Soto and Senf, 2025), and GAMI from
787 <https://doi.org/10.5880/GFZ.1.4.2023.006> (Besnard et al., 2024).

788

789 **Code availability**

790 The code used to produce the data products is available at
791 <https://doi.org/10.5281/zenodo.23503540> (Pellissier-Tanon, 2026) and
792 https://github.com/agnespeltan/essd_france_forest_code

793 **Author contributions**

794 APT and PC designed the study, with contributions from YX and FR on the growth-curve
795 framework. APT produced the data products, incorporating methodological input from YX and
796 FR. NB provided validation data. APT, PC, and NB contributed to the data analysis. PC
797 supervised the research. APT prepared the manuscript with contributions from all co-authors. All
798 authors contributed to the interpretation of the results and to manuscript revision.

799 **Competing interests**

800 The authors declare that they have no conflict of interest.



801 Acknowledgements

802 This work was granted access to the HPC resources of IDRIS under allocations 2025-
 803 AD010114718R1 and 2026-AD010114718R2 made by GENCI.

804 We used AI assistants to clean and organize analysis code and to improve the clarity of the text;
 805 the authors reviewed all outputs and remain fully responsible for the manuscript and software.

806 Financial support

807 This work was supported by the European Space Agency (ESA) project EO-LINCS (Earth
 808 Observation, Local and Integrated data eNquiry for Terrestrial Carbon Science), contract no.
 809 ESA 4000144915/24/I-EF, within the ESA Carbon Science Cluster (Research Opportunities 2 –
 810 Theme 2: Carbon Data Enquiry and Benchmarking), and by the AI4Forest project, funded by
 811 ANR (grant no. ANR-22-FAI1-0002) and BMBF (grant no. 01IS23025A).

812 References

- 813 Besnard, S., Koirala, S., Santoro, M., Weber, U., Nelson, J., Gütter, J., Herault, B., Kassi, J.,
 814 N'Guessan, A., Neigh, C., Poulter, B., Zhang, T., and Carvalhais, N.: Mapping global forest age
 815 from forest inventories, biomass and climate data, *Earth Syst. Sci. Data*, 13, 4881–4896,
 816 <https://doi.org/10.5194/essd-13-4881-2021>, 2021.
- 817 Besnard, S., Santoro, M., Herold, M., Cartus, O., Gütter, J., Heinrich, V. H. A., Herault, B.,
 818 Kassi, J., Koirala, S., N'Guessan, A., Neigh, C., Nelson, J. A., Poulter, B., Weber, U., Zhang, T.,
 819 and Carvalhais, N.: Global Age Mapping Integration (GAMI) (2.0),
 820 <https://doi.org/10.5880/GFZ.1.4.2023.006>, 2024.
- 821 Canopée: Observatoire des coupes rases – Rapport, Canopée – Forêts vivantes, Paris,
 822 <https://www.actu-environnement.com/media/pdf/news-47600-canopee-rapport-coupes-rases.pdf>
 823 (last access: 5 October 2026), 2026.
- 824 Cavaignac, S.: Les sylvoécotégions (SER) de France métropolitaine Étude de définition, Report,
 825 French National Forest Inventory, Nogent-sur-Vernisson, 2009.
- 826 Ciais, P., Zhou, C., Schneider, P., Schwartz, M., Besic, N., Vega, C., and Bontemps, J.-D.: An
 827 outlook on the rapid decline of carbon sequestration and perspectives for an improved
 828 monitoring of French forests, *Comptes Rendus Géoscience*, 358, 27–49,
 829 <https://doi.org/10.5802/crgeos.309>, 2026.
- 830 Forzieri, G., Girardello, M., Ceccherini, G., Spinoni, J., Feyen, L., Hartmann, H., Beck, P. S. A.,
 831 Camps-Valls, G., Chirici, G., Mauri, A., and Cescatti, A.: Emergent vulnerability to climate-



- 832 driven disturbances in European forests, *Nat. Commun.*, 12, 1081,
 833 <https://doi.org/10.1038/s41467-021-21399-7>, 2021.
- 834 Grünig, M., Rammer, W., Senf, C., Albrich, K., André, F., Augustynczyk, A. L. D., Baumann,
 835 M., Bohn, F. J., Bouwman, M., Bugmann, H., Collalti, A., Cristal, I., Dalmonech, D., De
 836 Coligny, F., Dobor, L., Dollinger, C., Espelta, J. M., Forrester, D. I., Garcia-Gonzalo, J.,
 837 González-Olabarria, J. R., Hiltner, U., Hlásny, T., Honkaniemi, J., Huber, N., Jonard, M.,
 838 Jönsson, A. M., Kunstler, G., Lagergren, F., Lindner, M., Mina, M., Moos, C., Morin, X., Muys,
 839 B., Nabuurs, G.-J., Nieberg, M., Patacca, M., Peltoniemi, M., Reyer, C. P. O., Schelhaas, M.-J.,
 840 Storms, I., Thom, D., Toïgo, M., and Seidl, R.: Climate change will increase forest disturbances
 841 in Europe throughout the 21st century, *Science*, 391, eadx6329,
 842 <https://doi.org/10.1126/science.adx6329>, 2026.
- 843 IGN: BD Foret® version 2, <https://geoservices.ign.fr/bdforet> (last access: 17 April 2023), 2018.
- 844 IGN: IGN – Inventaire forestier national français, Données brutes, Campagnes annuelles 2005 et
 845 suivantes, <https://inventaire-forestier.ign.fr/dataIFN/> (last access: 9 October 2025a), 2025.
- 846 IGN: Les coupes de bois, <https://observatoire.foret.gouv.fr/themes/les-coupes-de-bois>, last
 847 access: 28 January 2026b, 2025., 2025.
- 848 IGN: Mémento inventaire forestier IGN - 2025, IGN - National Forest Inventory of France,
 849 <https://www.calameo.com/read/0011885827c418636cbd4> (last access: 21 April 2026c), 2025.
- 850 IPCC: Special Report on Climate Change and Land, 2019.
- 851 Joshi, N., Mitchard, E. T. A., Brolly, M., Schumacher, J., Fernández-Landa, A., Johannsen, V.
 852 K., Marchamalo, M., and Fensholt, R.: Understanding ‘saturation’ of radar signals over forests,
 853 *Sci. Rep.*, 7, 3505, <https://doi.org/10.1038/s41598-017-03469-3>, 2017.
- 854 Kowalski, K., Viana-Soto, A., Brandt, M., Ciais, P., De Keersmaecker, W., Fensholt, R., Pugh,
 855 T. A. M., Xu, Y., and Senf, C.: Accelerating biomass loss from forest disturbances across
 856 Europe, *Nat. Geosci.*, 19, 1132–1138, <https://doi.org/10.1038/s41561-026-02032-y>, 2026.
- 857 Moreno, A., Neumann, M., and Hasenauer, H.: Forest structures across Europe, *Geosci. Data J.*,
 858 4, 17–28, <https://doi.org/10.1002/gdj3.45>, 2017.
- 859 Mutanga, O., Masenyama, A., and Sibanda, M.: Spectral saturation in the remote sensing of
 860 high-density vegetation traits: A systematic review of progress, challenges, and prospects, *ISPRS*
 861 *J. Photogramm. Remote Sens.*, 198, 297–309, <https://doi.org/10.1016/j.isprsjprs.2023.03.010>,
 862 2023.
- 863 Patacca, M., Grünig, M., Schelhaas, M.-J., Alberdi, I., Filipek, S., Fridman, J., Järvikylä, J.,
 864 Seidl, R., Senf, C., Staritsky, I., Zudin, S., Lohbeck, M., and Nabuurs, G.-J.: Climate-driven
 865 increases in wildfire projected to affect European forest types differently, *Environ. Res. Lett.*, 21,
 866 034027, <https://doi.org/10.1088/1748-9326/ae4115>, 2026.



- 867 Pellissier-Tanon, A.: `essd_france_forest_code`, Zenodo [code]
 868 <https://doi.org/10.5281/zenodo.23503540>, 2026.
- 869 Pellissier-Tanon, A., Ciais, P., Schwartz, M., Fayad, I., Xu, Y., Ritter, F., de Truchis, A., and
 870 Leban, J.-M.: Combining satellite images with national forest inventory measurements for
 871 monitoring post-disturbance forest height growth, *Front. Remote Sens.*, 5,
 872 <https://doi.org/10.3389/frsen.2024.1432577>, 2024.
- 873 Pellissier-Tanon, A., Xu, Y., Ritter, F., Besic, N., and Ciais, P.: Species- and region-specific
 874 growth curves, forest age, and disturbance trends for French forests at 30 m resolution (2020),
 875 Zenodo [dataset], <https://doi.org/10.5281/zenodo.23499335>, 2026.
- 876 Pilli, R., Blujdea, V. N. B., Rougieux, P., Grassi, G., and Mubareka, S. B.: Volume, increment,
 877 and aboveground biomass data series and biomass conversion and expansion factors for the main
 878 forest types of EU Member States, *Ann. For. Sci.*, 81, 35, [https://doi.org/10.1186/s13595-024-](https://doi.org/10.1186/s13595-024-01256-5)
 879 [01256-5](https://doi.org/10.1186/s13595-024-01256-5), 2024.
- 880 Poulter, B., Aragão, L., Andela, N., Bellassen, V., Ciais, P., Kato, T., Lin, X., Nachin, B.,
 881 Luyssaert, S., Pederson, N., Peylin, P., Piao, S., Pugh, T., Saatchi, S., Schepaschenko, D.,
 882 Schelhaas, M., and Shvidenko, A.: The global forest age dataset and its uncertainties
 883 (GFADv1.1), <https://doi.org/10.1594/PANGAEA.897392>, 2019.
- 884 Robinson, N., Drever, C. R., Gibbs, D. A., Lister, K., Esquivel-Muelbert, A., Heinrich, V., Ciais,
 885 P., Silva-Junior, C. H. L., Liu, Z., Pugh, T. A. M., Saatchi, S., Xu, Y., and Cook-Patton, S. C.:
 886 Protect young secondary forests for optimum carbon removal, *Nat. Clim. Change*, 15, 793–800,
 887 <https://doi.org/10.1038/s41558-025-02355-5>, 2025.
- 888 Sáez-Cano, G., Marvá, M., Ruiz-Benito, P., and Zavala, M. A.: Modelling Tree Growth in
 889 Monospecific Forests from Forest Inventory Data, *Forests*, 12, 753,
 890 <https://doi.org/10.3390/f12060753>, 2021.
- 891 Schwartz, M., Ciais, P., De Truchis, A., Chave, J., Ottlé, C., Vega, C., Wigneron, J.-P., Nicolas,
 892 M., Jouaber, S., Liu, S., Brandt, M., and Fayad, I.: FORMS: Forest Multiple Source height, wood
 893 volume, and biomass maps in France at 10 to 30 m resolution based on Sentinel-1,
 894 Sentinel-2, and Global Ecosystem Dynamics Investigation (GEDI) data with a deep learning
 895 approach, *Earth Syst. Sci. Data*, 15, 4927–4945, <https://doi.org/10.5194/essd-15-4927-2023>,
 896 2023.
- 897 Schwartz, M., Fogel, F., Besic, N., Robert, D., Geist, L., Renaud, J.-P., Monnet, J.-M., Mosig,
 898 C., Vega, C., d’Aspremont, A., Landrieu, L., and Ciais, P.: FORMSpOT: Revealing fine-scale
 899 forest disturbances from nation-wide 1.5 m forest canopy height time series, *Remote Sens.*
 900 *Environ.*, 346, 115631, <https://doi.org/10.1016/j.rse.2026.115631>, 2026.
- 901 Shchur, A., Bragina, E., Sieber, A., Pidgeon, A. M., and Radeloff, V. C.: Monitoring selective
 902 logging with Landsat satellite imagery reveals that protected forests in Western Siberia
 903 experience greater harvest than non-protected forests, *Environ. Conserv.*, 44, 191–199,
 904 <https://doi.org/10.1017/S0376892916000576>, 2017.



- 905 Viana-Soto, A. and Senf, C.: The European Forest Disturbance Atlas: a forest disturbance
906 monitoring system using the Landsat archive, *Earth Syst. Sci. Data*, 17, 2373–2404,
907 <https://doi.org/10.5194/essd-17-2373-2025>, 2025.
- 908 Vilén, T., Gunia, K., Verkerk, P. J., Seidl, R., Schelhaas, M.-J., Lindner, M., and Bellassen, V.:
909 Reconstructed forest age structure in Europe 1950–2010, *For. Ecol. Manag.*, 286, 203–218,
910 <https://doi.org/10.1016/j.foreco.2012.08.048>, 2012.
- 911 Wang, Y., Wang, H., Liang, C., Li, X., Liu, Z., Zhang, X., Li, S., Yang, J., and Zhao, Y.: A
912 global 30 m disturbance-recovery age dataset for young natural and planted forests (1985–2024),
913 *Earth Syst. Sci. Data*, 18, 6225–6249, <https://doi.org/10.5194/essd-18-6225-2026>, 2026.
- 914 Zeng, W., Wen, X., Sun, X., Yang, X., Pu, Y., and Zhang, L.: Developing Biomass Growth
915 Models for Chinese Fir Plantations Based on National Forest Inventory Data, *Forests*, 17, 485,
916 <https://doi.org/10.3390/f17040485>, 2026.