

Article

Baseline Mortality Analysis Reveals Legacy of Contrasting Land Use Practices on the Structural Sustainability of Endangered Moroccan and Spanish Mountain Forests

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Abstract: Disturbances can affect forest health and are important modulating factors of tree responses to environmental changes. However, standard methods are needed to assess and elucidate the relative effects of disturbance legacies on forest health among species. Here, structural sustainability was used to evaluate and to compare the impacts of contrasting disturbances on the health of *Quercus ilex*, *Cedrus atlantica*, and *Abies pinsapo* forest stands in Morocco and Spain. Disturbance effects on structural sustainability were related to type, severity, and land-use history, and showed inter-regional variability. *Cedrus atlantica* was structurally sustainable in its core distribution in Morocco, but not at its southern and northernmost geographical range limits. *Quercus ilex* was structurally sustainable in Morocco. *Abies pinsapo* was structurally sustainable at optimal elevation sites in Morocco (Rif Mts.), but considering the whole *A. pinsapo* dataset including Morocco and Spain, the species is structurally unsustainable due to excessive mortality. However, at the lower elevation plots in Spain, unsustainability was due to insufficient mortality. Although some forests were structurally sustainable, none of them were deemed healthy because none met their management objectives. Results also support the key role of disturbance regimes as drivers of forest structural sustainability and adaptive capacity.

Keywords: climate change; drought; forest health; anthropogenic disturbance

1. Introduction

Forest structure is a driving force behind forest growth processes and productivity [1]. Forests worldwide are impacted by both natural and anthropogenic disturbances including forest fragmentation, improper logging practices, overgrazing, insect and disease outbreaks, air pollution, and drought among others. Substantial, long-term disturbance impacts result from changes in spatial forest structure [1]. The impact of these disturbances may surpass the resistance—the capacity to endure disturbance without significant community change—or resilience—community recovery from disturbance—of forests, resulting in substantial mortality or even compositional or structural shifts over time. Changes in forest structure and land use history, in particular, can have cascading, negative effects on co-dependent forest species such as wildlife and vegetation [2], as well as on ecosystem functions and services (e.g., primary productivity, [3], carbon accumulation [4]). How these functions respond to mortality likely will link to forest structure as a legacy of forest management. Indeed, recent

studies including managed stands and unmanaged by recent abandonment, concluded that the negative effect of climate warming on forest growth disappears if only managed stands are considered, emphasizing the potential of forest structure management to mitigate the effects of climate change [5].

Despite forest health, a complex concept that has proven difficult to define [6], the structural sustainability of a given forest might be used to evaluate overall forest health. Recently, many varied structural indices have been developed to quantify forest structure that may be useful as surrogate measures to quantify biotic aspects of forest health [7]. Castello and Teale [8] recently proposed a healthy forest as one that both meets landowner objectives (i.e., is productive) and is structurally sustainable (i.e., is sustainable). Structural sustainability can be defined as, at a single point in time, the degree to which mortality and development (i.e., regeneration/ingrowth) are comparable such that living trees have sufficient access to resources [8]. This is evaluated at the landscape-level to aggregate smaller-scale heterogeneity in forest characteristics based on site-specific factors [8]. The concept of structural sustainability [8,9] predicts that biotic and abiotic factors are more stable in more structurally sustainable forests. While the dynamics of uneven-aged forests are tied to a heterogeneous environment created by relatively low-impact, frequent mortality events, relatively large mortality events can cause significant deviations between forest mortality and development. The cascading impacts on forest ecosystems can be evaluated using baseline mortality analysis (BMA) [9], which compares current mortality levels to a reference level (i.e., baseline) derived from a distribution of live trees.

Expanding the quantitative nature of BMA, Cale et al. [9] described an index of structural sustainability, which has been used to identify as well as to discriminate between forests (or tree species) with (“structurally unsustainable”) or without (“structurally sustainable”) significant live-dead deviations. However, this index is not designed to predict a static estimate or classification over time, but instead to estimate structural sustainability for a single point in time. Further, while this index has been used to compare the relative unsustainability/sustainability among species [9], comparisons among species experiencing similar disturbances have not yet been assessed.

Resilience and stability of forest ecosystems often are linked to structural and species diversity levels [1,2,10]. An understanding of interdependencies among biotic, abiotic, and anthropogenic disturbance impacts on forest structure is important to adapting forest management and conservation tactics, especially for endangered tree species, to those impacts [11]. An understanding of the relationship between structural sustainability and disturbance impacts also could bolster forest triage and monitoring efforts by allowing forest managers to compare disturbance impacts, and their potential interactions, among sites and over time [12].

Here we used the index developed by Cale et al. [9] to compare the structural sustainability of variously disturbed stands of forest trees in Morocco (*Abies pinsapo*, *Quercus ilex*, and *Cedrus atlantica*), and Spain (*A. pinsapo*). We seek to assess whether the history, type, and/or severity of these disturbances are reflected in the structural sustainability and health of the tree populations examined.

2. Experimental Section

2.1. Materials and Methods

2.1.1. Sampling Locations, Site Characteristics, Disturbance Histories, and Datasets

Table 1 lists the scientific and common names, sampling site locations, total number of trees and area sampled as well as mean plot size by region based on elevation.

Abies pinsapo forests are areas of high biodiversity and endemism. It is a relict species of a group of circum-Mediterranean firs. It grows in small and isolated populations between 1000–1800 meters above sea level (ASL) on north-facing slopes and ravines of some coastal mountain ranges of southern Spain (west Betic Range, approximately 3600 ha) and northern Morocco (Rif Mts, approximately 3000 ha) [5]. These forests were intensely damaged over the centuries through a series of unsustainable practices including overgrazing, uncontrolled logging, and pollarding [13]. In southern Spain, these forests expanded in size and density following conservation practices in the mid-20th century [13].

Some of the sampling sites at low elevation in southern Spain are dense, uneven-aged forests with much past degradation and drought, as well as root rot and beetle damage in parts of it, while other parts are non-anthropogenically disturbed, stem exclusion stage forests that have been well protected. The high elevation sites are old growth, well preserved forests without high-impact disturbances.

Table 1. Scientific and common names, sampling locations, number of trees sampled, total area sampled, and mean plot size for each of the populations.

Species	Common Name	Country, Region	No. of Trees Sampled	Total Area Sampled (ha)/Mean Plot Size (ha)
<i>Abies pinsapo</i> Boiss.	Mediterranean fir	Spain and Morocco	3405	6.3/0.15
		Spain	2835	3.89/0.158
		Spain, low elevation	1830	1/0.066
		Rif Mts.	573	1.38 ha/0.15
<i>Cedrus atlantica</i> (Endl.) Manetti ex Carriere	Atlas cedar	Morocco	3320	6.6/0.24
		High Atlas Mts.	1378	2.8
		Mid-Atlas Mts.	940	1.42
		Rif Mts.	607	0.93
<i>Quercus ilex</i> L.	Holm oak	Morocco	1098	5.0/0.21

The fir stands of the Rif Mts of northern Morocco were subjected to continued forest harvest, high human population density, and both legal and illegal forest exploitation (e.g., cutting of small trees for firewood, pollarding for forage) that resulted in loss of natural vegetation cover [14].

Abies pinsapo forests are included in the Mediterranean intercontinental Biosphere Reserve of Spain and Morocco established by UNESCO's Man and the Biosphere Program [5]. Research studies on these endangered tree species focus on environmental conditions, regeneration status, and causes of decline [5]. Structural studies to identify the effects of past human impacts, recent land changes, and climate change impacts on the viability of such endangered species have not been done [5]. *Abies pinsapo* field data were collected between 2003–2007 in southern Spain and Morocco. Soils were mainly mesic oxiaquic hapludols or mesic calcic haploexerepts [5].

The *Cedrus atlantica* and *Q. ilex* sampling sites were located in the Rif, Middle, and High Atlas Mts of northern Morocco. In the Rif Mts they occur at the highest elevation in zone 1 at 1800 to 2000 m ASL [14]. The characteristics of this region are similar to that described above for *A. pinsapo*. At these elevations (1800–2000 m) the overstory is *C. atlantica* and the understory is *Q. ilex*. The climate is humid and cold. Soils are cambisols. Slopes are steep, and roughland grazing (goats) is a principal disturbance. The cedar here were lightly logged then protected and managed as relict cedar forests.

Cedrus atlantica occurs between 1000 and 1800 m ASL in the Mid-Atlas Mts., but is rare below 1200 m due to land use pressure. Grazing pressure is high [15]. Cedar occurs mixed with *Q. ilex*, which was abundant in the sampled sites [16]. Recurrent moderate droughts and temperature increases have occurred since the 1970s [17]. The Mid-Atlas Mts. contain about 80% of the entire *C. atlantica* forest surface area (100,000 ha). Dieback and decline are common especially in the old trees, and where *Q. ilex* is most abundant in the understory and exhibits phenotypic plasticity and high adaptive capacity to environmental stress [18,19] (Figure 1).

The High Atlas Mts. represent the southern-most geographical range limit of *C. atlantica*. This species has been severely degraded in this region because of its drought sensitivity, proximity to the Sahara desert, and history of human disturbance and severe recent drought events [17]. The High Atlas Mts. only receive 500 mm of rain per year. The region was intensively logged, experienced severe overgrazing by goat herds and pollarding to provide winter forage for goats.



Figure 1. Mortality of old Atlas cedar (*Cedrus atlantica*) in the Middle Atlas Mts. (North Morocco, Photo from J.C. Linares).

2.1.2. Baseline Mortality and Structural Sustainability Index.

Each dataset (Table 1) contained the number and diameter at breast height (dbh) of every live and dead tree of the species of interest on the sampled plots. Average growth rates for each species were either calculated from radial growth data or were obtained from regional online databases. The time period that dead trees of a given species remained identifiable in the field was estimated through discussion with foresters, or others familiar with the species in the specific region. The diameter class (DC) bin size for both living and dead trees was then determined by multiplying the mean growth rate (cm/year) by the time required for dead trees to become unrecognizable due to decay (year). This was done to place dead, decaying trees into the diameter class of their living cohort members, and, in so doing, help make baseline mortality and observed mortality estimates comparable [8].

Baseline mortality analyses and the structural sustainability index of Cale, et al. [9] were applied to populations of *C. atlantica*, *A. pinsapo* and *Q. ilex* in Morocco, and *A. pinsapo* in Spain.

Specifically, the baseline mortality of each species population was calculated from the diameter distribution of the living trees according to Castello et al. [8] and Cale et al. [9]. This process involved first using linear regression to calculate a baseline-level of mortality from the diameter distribution of log-linear counts of living trees, then statistically comparing, by chi-squared tests, observed dead and expected dead (i.e., number dead calculated from baseline mortality) tree densities in series for each DC [8]. These analyses were done using the freely available Forest Sustainability Calculator software (<http://www.esf.edu/efb/forsustcalc/>) [9]. This software further classifies a given diameter distribution as “sustainable” or “unsustainable” using a structural sustainability index [9].

While specific details are provided in Cale et al. [9], we briefly describe structural sustainability index score calculation here. These scores are calculated using a linear discriminant function (Equation (1)) comprised of values of five metrics characterizing BMA results: (1) aggregation (extent of clustering among DCs with significant differences; AGG), (2) relative abundance (proportion of DCs with significant differences; RA), (3) magnitude (total difference between observed and baseline mortality among all DCs exhibiting differences; MAG), (4) distribution (where differences are located in the DC range; DM), and (5) change (absolute difference in the number of DCs with significant differences in the first and last distribution of an iterative simulation; CHG) [9]. The resulting index scores can be used to assess the relative structural unsustainability of diameter distribution. Further, index scores can be used to classify distributions as structurally “sustainable” or “unsustainable” based on whether they are greater than or less than, respectively, a threshold score of 70.6 [9].

$$\text{Score} = (0.699) \text{ AGG} + (0.684) \text{ RA} + (0.535) \text{ MAG} + (0.420) \text{ DM} - (0.554) \text{ CHG} \quad (1)$$

3. Results

The species evaluated showed negative exponential diameter distributions for living trees, which, following log-transformation, had significant ($p < 0.05$) negative relationships with diameter class with R^2 of 0.69–0.97 (Table 2). Six of the nine datasets we examined had a log-linear OLS negative exponential regression model with an R^2 value > 0.8 . Cale et al. [9] assumed that such models satisfactorily fit the diameter distribution of that species population. Three of our species populations (*A. pinsapo* in the Rif Mts, and *C. atlantica* in the High Atlas and Rif Mts of Morocco), however, did not fit this assumption (Table 2). Baseline mortality differed among species and populations, ranging from 14.8% to 33.0% (Table 2). Output of Forest Sustainability Calculator program showing those diameter classes with significant differences between observed and expected mortality based on chi squared analysis for the datasets examined is presented in Table 3.

Table 2. Baseline percent mortality, R^2 of regression of natural log of diameter distribution fit to the negative exponential model, index score, and sustainability category (sustainable [S] or unsustainable [U] due to excessive [+] or insufficient [−] mortality) for each of the species populations by sample region.

Species	Region	Baseline Mortality (%)	R^2	Index Score	Sustainability Category
<i>Abies pinsapo</i>	Spain and Morocco	14.8	0.88	73.16	U, +
	Spain	14.8	0.88	73.56	U, +
	Spain, low elevation	33	0.95	89.33	U, −
	Rif Mts.	18.1	0.76	9.63	S
<i>Cedrus atlantica</i>	Morocco	22.1	0.96	161.76	U, +
	High Atlas Mts	18.1	0.69	245.48	U, +
	Mid-Atlas Mts	22.1	0.97	25.86	S
	Rif Mts	18.1	0.72	93.34	U, +
<i>Quercus ilex</i>	Morocco	17.1	0.90	15.79	S

Table 3. Baseline mortality analysis results showing the diameter classes with significant differences between observed and expected mortality based on chi squared analysis for all datasets examined.

Significant Diameter Classes (cm) by Species and Plot	Observed Dead	Expected Dead	Chi Square (Critical Value)
<i>Abies pinsapo</i> (all Spain & Morocco plots)			
4	136	87.97	26.22 (8.95)
8	131	94.77	13.85
44–48	20	10.34	9.00
52	14	5.77	11.76
64	14	5.47	13.30
68–72	20	12.56	16.59
<i>A. pinsapo</i> (all Spain plots)			
4	135	79.1	39.50 (8.84)
8	115	70.53	28.04
44	12	5.17	9.00
56	17	7.97	10.18
64	22	7.98	24.60
<i>A. pinsapo</i> (Spain low elevation plots)			
8	82	118.68	11.34 (8.05)
12	25	111.1	66.73
16	26	90.0	45.51
20	12	65.94	44.12
24	14	34.29	12.00
28	11	26.37	8.96
32	1	14.18	12.25
<i>A. pinsapo</i> (all Morocco plots)			
4	3	15.04	9.64 (7.48)
<i>Cedrus atlantica</i> (all plots)			
5	70	202.83	86.99 (8.95)
10	201	114.13	66.10
15	226	102.636	148.27
20	116	71.67	27.42
85–95	19	6.79	21.50
100–110	12	6.41	10.26
<i>C. atlantica</i> (High Atlas Mts)			
5	4	137.58	129.69(8.21)
15	43	13.59	63.59
20	33	11.24	42.13
25	22	7.25	30.00
35	14	5.07	15.69
40–45	28	9.06	39.56
50–55	19	5.79	30.03
85–100	14	5.07	15.69
<i>C. atlantica</i> (Mid-Atlas Mts)			
10	103	45.12	74.23 (8.05)
15	74	44.02	20.42
<i>C. atlantica</i> (Rif Mts)			
10	75	23.02	117.36(7.48)
15	106	25.02	262.18
20	50	15.41	77.66
25	26	11.06	20.19
<i>Quercus ilex</i> (Morocco)			
3.75	5	26.329	17.28(8.36)
6.25	3	16.054	10.62

Regional differences in structural sustainability were observed for *C. atlantica*, and *A. pinsapo* (Tables 2 and 4). The combined population of *A. pinsapo* in Spain, and the combined populations in Spain and Morocco were structurally unsustainable due to excessive mortality (scores of 73.6 and 73.2,

respectively, Tables 2 and 4). The *A. pinsapo* population in Morocco (the Rif Mts) was structurally sustainable (score of 9.6, Tables 2 and 4), whereas the low elevation mountain stands in Spain were unsustainable due to insufficient mortality (score of 89.3, Tables 2 and 4). The combined population of *C. atlantica* in Morocco was structurally unsustainable due to excessive mortality (score of 161.7, Tables 2 and 4). The regional populations in the High Atlas Mts and the Rif Mts were structurally unsustainable due to excessive mortality with scores of 245.5 and 93.3, respectively, Tables 2 and 4). However, the *C. atlantica* population in the Middle Atlas Mts was structurally sustainable (score of 25.9, Tables 2 and 4). *Quercus ilex* also had a structurally sustainable diameter distribution in Morocco (Tables 2 and 4).

Table 4. Diameter metric values and sustainability scores (calculated using Equation (1)) for *Abies pinsapo*, *Cedrus atlantica*, and *Quercus ilex* populations in several regions of Spain and Morocco.

Structural Sustainability Index Metrics							
Species	Region	Aggregation	Relative Abundance	Magnitude	Distribution of Mortality	Change	Index Score
<i>Abies pinsapo</i>	Spain and Morocco	3	0.30	125.26	10	0.63	73.16
	Spain	3	0.26	126.05	10	0.63	73.56
	Spain, low elevation	3	0.47	155.52	9	0.13	89.33
	Rif Mts.	0	0.05	14.51	5	0.49	9.63
<i>Cedrus atlantica</i>	All regions	5	0.48	290.24	7	0.50	161.76
	High Atlas Mts.	5	0.65	440.97	14	0.47	245.48
	Mid. Atlas Mts.	1	0.13	43.44	5	0.49	25.86
	Rif Mts.	2	0.33	164.81	9	0.43	93.34
<i>Quercus ilex</i>	Morocco	0	0.08	26.06	5	0.55	15.79

4. Discussion

Most temperate forests are accumulating carbon and may potentially act as carbon sinks in the near future [18]. However, the situation may be different in water-limited ecosystems, such as *A. pinsapo* and *C. atlantica* forests [5,15], where the potentially positive effects of rising atmospheric CO₂ and temperature will likely interact with water availability [20]. As regards modulating factors of forest growth under a warming climate, forest structure should be taken into account as a likely driver of future stand sustainability [11]. Two of the three disturbance-impacted species that we evaluated here had structural sustainability problems, which varied by sampling region (Tables 2–4). Therefore, not all of the disturbances having impacted these forests resulted in unsustainable forest structures, which likely reflects inherent differences in tree species resilience. Differences in regional structural sustainability scores also agree with variation in regional drought severity and anthropogenic disturbance history, type, and severity. At regional scales, water availability was positively related to forest growth, while recent warming has reduced growth rate especially in drought-prone areas [15,16,21]. *Cedrus atlantica* in the High Atlas Mts of Morocco, for example, had the highest unsustainability score (score of 245.48, Tables 2 and 4). Because *C. atlantica* occurs from early to later successional stages and is without stand-replacing biotic mortality-agents, the detected unsustainability is not likely explained by population senescence or pests [22]. Therefore, this score reflects the impacts of known region-wide severe drought events and anthropogenic disturbances, manifesting as excessive mortality in most diameter classes (Table 3). This region receives an average of only 500 mm of rain/year [23], and overgrazing and pollarding to provide winter forage for goats are prevalent here [18]. Furthermore, intensive timber extraction by logging in the mid-20st century, as well as long-term trends of climate dryness further degraded these forests, likely acting as contributing factors of recent *C. atlantica* dieback and mortality [16,24–26].

However, at the regional-level, these impacts have variously interacted with forest management-associated disturbances, such as past degradation from logging (e.g., in the Atlas Mts.) [25]. *Cedrus atlantica* forests in the Middle Atlas Mts also experienced severe human disturbances, but the drought was less severe here than elsewhere in Morocco (about 1000 mm precipitation/year, [23]). These forests remain structurally sustainable (Tables 2 and 4) despite a history of moderate droughts [26,27] causing heavy dieback and mortality of older trees [17]. (Figure 1). *Cedrus atlantica* however, was structurally unsustainable near its northern distribution limit (i.e., the Rif Mts of Morocco; score of 93.34, Tables 2 and 4) even though drought stress is not significant here (more than 1800 mm precipitation/year) [23], and human disturbance is not as severe as in the Middle and High Atlas [28]. These results suggest a recent expansion of *C. atlantica* at its northern geographical range limit, which is currently related to a lack of size and age diversification and reduced stand mortality. Conversely, ongoing decline of *C. atlantica* stands in intensively logged and overgrazed areas of the Middle Atlas (Figures 1 and 2), although structurally sustainable, are leading to their replacement by *Q. ilex* [16] (Figure 2), which also was classified here as structurally sustainable (score 15.79, Tables 2 and 4). Therefore, high-frequency cutting and worsening drought events are encouraging the expansion of high-density drought-tolerant *Q. ilex* as the *C. atlantica* overstory continues declining. This development will likely hasten dominant species replacement in this region [16].



Figure 2. Decline symptoms observed in Atlas cedar (*Cedrus atlantica*) forests from Middle Atlas (North Morocco, Photo from V. Lechuga). (Note that *Quercus ilex* is the broadleaf species that replaces Atlas cedar following mortality of canopy trees).

Recent studies have found an interaction gradient between pastoral and woodcutting activities, where the disturbance having the most impact on *C. atlantica* stand structure is grazing and pruning [29]. This study concludes that the structure differences depend more on human activities than on environmental parameters. Unexpectedly, altitude was not an important factor neither explaining *C. atlantica* forest structure nor in the prevailing land-use type [20]. According to our results, it is suggested that at regional scales, the J-shaped *C. atlantica* and *Q. ilex* size class structures found in the Middle Atlas region could be interpreted as dynamic and healthy populations, while it is not at its southern and northernmost geographical range limits (Tables 3 and 4).

Abies pinsapo in Spain overall as well as in Spain and Morocco combined was structurally unsustainable due to excessive mortality (score of 73.56, and 73.16, respectively, Tables 2 and 4). In Morocco, however, this species was structurally sustainable (score of 9.63, Tables 2 and 4), while the lower elevation Spanish plots showed unsustainability due to insufficient mortality (score of 89.33, Tables 2 and 4). This was not observed at other Spanish stands where the species was unsustainable due to excessive mortality (score of 73.56. Tables 2 and 4). The sustainability index indicates disturbance-induced mortality has resulted in unsustainable *A. pinsapo* forests in Spain, but not in Morocco (Tables 2 and 4). Spatial variation in mortality might be explained by differences in growth rate across plots [13]. Here, we illustrate that forest structural variables, as a legacy of previous forest management, are among the main determinants of forest growth and, consequently, carbon stock trends. Structural patterns observed in *A. pinsapo* from Spain may be consistent with the expected self-thinning dynamics related to the recent abandonment of forest management, with the consequent increase in tree density and competition [13,21].

Currently undisturbed *A. pinsapo* forests in Spain are growing, and likely accumulating carbon, at a lower rate than those subjected to low-intensity disturbances by local inhabitants in Morocco [17]. Between these regions in Spain, *A. pinsapo* forests show a multilayered canopy and a strong dependence for tree recruitment on gap formation and light availability [13]. Many of these forests were intensely damaged over centuries of various unsustainable practices such as overgrazing, unregulated logging, and pollarding [17]. Following conservation efforts and progressive abandonment of traditional practices in the 1950s, fragmented tree populations expanded and the scattered remaining stands became denser [13]. This created a landscape mosaic of forests with various land-use histories and structures from which patterns of stand structural dynamics of *A. pinsapo* can be reconstructed.

Because *A. pinsapo* is a late-successional species without cyclically-occurring pests, events such as population senescence and biotic agent outbreaks do not explain the detected structural unsustainability [13]. Thus, the results here likely reflect differences in this land use history (Tables 2–4). For example, early-aggradative, stem-exclusion stage forests established after protection policies were implemented were structurally unsustainable due to insufficient mortality (i.e., low-elevation stands in Spain, Tables 2 and 4). Conversely, other *A. pinsapo* forests in Spain having experienced heavy disturbance-induced mortality [13] showed unsustainability due to excessive mortality (Tables 2 and 4, all Spain AP regions combined but consisting mostly of trees in high elevation plots). These stand structures can be matched to known changes in forest-use as several tree-size classes are absent from the canopy, suggesting a history of intermittent heavy grazing by domestic animals that precluded tree establishment during certain periods [17].

Furthermore, in the last twenty years, increased canopy closure and potential climatic warming might explain a generalized tree-growth reduction in the low elevation *A. pinsapo* forest, related to canopy closure and high intra-specific competition [13]. Drought stress also was more severe in the low elevation stands than in the other stands [30]. In Morocco, however, early conservation/protection measures were not readily implemented. Therefore logging of small trees by locals, grazing, and pollarding continued to the present [17]. It is likely that the Spanish conservation measures led to the insufficient mortality experienced in the low elevation stands of Spain (Figure 3), which led to excessive tree competition as well as stand dieback and mortality attributed to root rot and bark beetles (Figure 4), supporting our belief that stand diversification and mortality are needed to modulate relict *A. pinsapo* forest vulnerability (Tables 2–4). The higher human disturbance pressure in the Rif Mts of Morocco probably prevented similar unsustainability in the Mid-Atlas Mts of Morocco (Tables 2–4), reflecting differences in disturbance history, type, and severity.



Figure 3. Photographs from 1929 (upper figure, photograph taken from Barbey, 1931) and 2003 (lower figure; Photo from J.C. Linares) at south Spain (Sierra de Grazalema) showing changes in *A. pinsapo* forest coverage following land use changes following *A. pinsapo* forest conservation.



Figure 4. Decline symptoms observed at lower elevation in dense *Abies pinsapo* forests from south Spain (Photo from J.C. Linares).

5. Conclusions

The results presented here illustrate that disturbance-associated mortality or the lack thereof does not necessarily create unsustainable forest structures. Instead, tolerant tree species populations may maintain a sustainable structure (e.g., *Q. ilex*), or disturbance may result in size-specific

changes in mortality causing unsustainability (e.g., *C. atlantica* all plots) or insufficient mortality (e.g., low-elevation *A. pinsapo* sites in Spain) potentially during pulses of post-disturbance growth. This variability highlights the importance of evaluating and comparing forest stand structures using similar metrics, (i.e., BMA and the structural sustainability index). We have used the index of Cale et al. [9] to compare the relative impacts of a specific disturbance or set of disturbances among endangered forest trees, in different regions. This index can further be used to aid environmental triage and monitoring programs to make quantitatively supported resource allocation decisions and to evaluate forest health over time, especially under novel disturbance pressures [12,31]. According to the two-part definition of a healthy forest as proposed by Castello and Teale [8], structural sustainability and satisfying landowner objectives (e.g., productivity, conservation, etc.) are integral factors of defining forest health. None of the forests assessed here currently meet the conservation management objectives mandated by their respective governing bodies [32]. Therefore, by definition, none of the structurally unsustainable forests (Tables 2 and 4) can be considered healthy. Although some of the forests studied are structurally sustainable (*C. atlantica* in the Middle Atlas and *Q. ilex* in Morocco, Tables 2 and 4) they similarly cannot be considered healthy. We conclude that legacies of site-specific disturbances leave detectable signals in the structural sustainability of forest trees decades after occurrence.

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