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A Quantitative Index of Forest Structural Sustainability

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Abstract: Forest health is a complex concept including many ecosystem functions, interactions and values. We develop a quantitative system applicable to many forest types to assess tree mortality with respect to stable forest structure and composition. We quantify impacts of observed tree mortality on structure by comparison to baseline mortality, and then develop a system that distinguishes between structurally stable and unstable forests. An empirical multivariate index of structural sustainability and a threshold value (70.6) derived from 22 nontropical tree species' datasets differentiated structurally sustainable from unsustainable diameter distributions. Twelve of 22 species populations were sustainable with a mean score of 33.2 (median = 27.6). Ten species populations were unsustainable with a mean score of 142.6 (median = 130.1). Among them, *Fagus grandifolia*, *Pinus lambertiana*, *P. ponderosa*, and *Nothofagus solandri* were attributable to known disturbances; whereas the unsustainability of *Abies balsamea*, *Acer rubrum*, *Calocedrus decurrens*, *Picea engelmannii*, *P. rubens, and Prunus serotina* populations were not. This approach provides the ecological framework for rational management decisions using routine inventory data to objectively: determine scope and direction of change in structure and composition, assess excessive or insufficient mortality, compare

disturbance impacts in time and space, and prioritize management needs and allocation of scarce resources.

Keywords: baseline mortality; climate change; forest health; invasive species

1. Introduction

The search for an objective definition of forest health and a framework for its assessment have been elusive [1]. Several definitions of forest health have been proposed that range from utilitarian to ecological in perspective [2]. Recently, Teale and Castello [3] proposed a two-part definition of forest health. The first is maintenance of a sustainable size class structure, within the ecological parameters of the species, at the landscape level as defined by Manion and Griffin [4]. The second involves meeting the management objectives for the specific forest. Sustainability, however, occupies a pivotal role because long-term management objectives become unattainable if the forest is unsustainable. Recent definitions of forest health incorporate important ecological qualities such as ecosystem balance, resilience to disturbance, plant and animal community function, sustainable productivity, as well as national, political, social, or spiritual concerns [1,5]. These factors are included in the management objective component of our definition of forest health. This study focuses on the sustainability aspect of the assessment of a healthy forest.
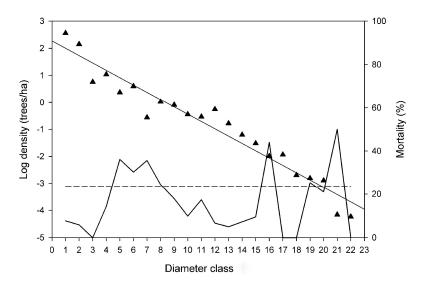
How does one define forest sustainability? Like forest health, sustainability has been difficult to define and to measure because people have differing and often competing priorities that reflect their values at a particular point in time and often come at the expense of others. For example, stakeholders (e.g., landowners, government agencies and forest ecologists) often disagree on what specific resources should be sustained and how forests should be managed. Furthermore, forest ecosystems are complex and develop slowly, confounding measures of sustainability [6]. The more traditional ecological concept of forest sustainability is based on the maintenance of forest ecosystem composition, structure, and function over the long term [2,7]. These approaches embody what many intuitively envision as a healthy forest, but are subjective, difficult to assess or both. Manion and Griffin [4] and Didion *et al.* [8] reasoned that stand age class structure be used to compare the combined effects of human and natural disturbances at the landscape scale over long time periods to assess the long-term sustainability of a forest. Didion *et al.* [8] utilized a simulation model-sensitivity analysis of forest age structure to evaluate the impacts of management regimes and fire on boreal forest sustainability. In principle, this approach is consistent with the diameter class structure approach of Manion and Griffin [4] and Zhang *et al.* [9] who used diameter as a surrogate for age [10].

We support an uncomplicated definition of forest sustainability simply as structural stability, and fully recognize that forests are dynamic, not static ecosystems as our definition may infer. We do not imply that forest structure or composition must remain unchanged for that forest to be considered sustainable or indeed, healthy. Rather it is the equivalence of baseline and observed mortality that must remain stable for the forest to be considered structurally sustainable. This approach gives us a frame of reference within which to evaluate changes in structure. It gives us the ability to objectively determine the scope and direction of change in structure and composition, which provides the ecological

framework for objective management decisions. It sets the stage for the quantification of forest stability or change that hinges on a comparison of the current, observed mortality with a context-specific, theoretical value termed baseline mortality. The calculation of current baseline mortality values are independent of prior BM values. Therefore, it is irrelevant whether forests are managed or not because the approach we developed applies to both systems.

The comparison of baseline to observed mortality is possible due to the q-ratio or its special case known as the Law of de Liocourt that describes the size structure of a forest (density of stems) as a function of stem diameter [11,12]. The q ratio applies only to uneven-aged stands; but since this is a landscape-level analytical approach, a forest consisting of a mosaic of even-aged stands can be uneven-aged at this scale [10]. Commonly, but not always, this relationship is described by the negative exponential function. When the diameter distribution closely fits the negative exponential function, the slope of the line is inversely proportional to the number of trees that must be removed from the population as the surviving trees in each diameter class grow into the next larger diameter class—for the size structure of the forest to remain stable. The baseline mortality is derived from this slope, and for diameter distributions that fit the negative exponential function, it is constant across all diameter classes (Figure 1). The method we describe potentially can be modified accordingly in instances where other probability density functions more closely approximate diameter distributions [9].

Figure 1. Log-linear plot of a hypothetical diameter distribution (▲), the baseline mortality calculated from the log-linear plot (- - -), and hypothetical currently observed mortality by diameter class (solid line without triangles).

Baseline mortality is the percent of trees in each diameter class that will normally die in a structurally sustainable forest in the time it takes for the trees to grow into the next larger diameter class. The width of the diameter classes, therefore, must correspond to the time frame in which the observed mortality occurred in order to provide a basis for the evaluation of mortality from all causes. For example, if dead stems are identifiable and measureable for a maximum of 20 years, then the diameter class width (DC bin size) in a baseline mortality analysis must represent 20 years of growth. The subjects of this paper are the precise relationship between observed and baseline mortality and

how much observed mortality must deviate from the baseline before we can expect a change in the size structure of the forest.

Here, we use available forest inventory datasets to assess the structural sustainability of 22 tree species in disparate geographic regions using the baseline mortality approach. We hypothesize that the structural sustainability of forest tree species populations and the impacts of known and unknown disturbances on their sustainability can be objectively and quantitatively assessed. Our objectives were to develop a quantitative index using a statistical approach to objectively distinguish between sustainable and unsustainable diameter distributions for 22 species and populations, and to evaluate the impacts of known disturbances on four of them. We used logistic regression to provide an unbiased threshold estimate. All sustainability scores above and below this estimate were deemed unsustainable and sustainable, respectively. We also developed a computer software package that performs all of the calculations necessary to conduct these analyses, and it is available on the SUNY-ESF website.

2. Experimental Section

2.1. Data

The datasets of 22 forest tree species were obtained from New Zealand (NZ) and the United States in this study. Table 1 lists the scientific and common names, sampling site locations, number of trees in each dataset, the total area sampled for each of the 22 species populations, and the mean plot size for each sampling location. Among them, the Adirondack Forest Preserve (AFP), New Zealand (NZ) and Alaska (AK) datasets were used in Castello *et al.* [13].

Table 1. Scientific and common names, sampling site locations, number of trees, total area sampled, and mean plot size for each of the 22 species populations utilized for baseline mortality analysis.

Species	Common name	Region [1]	No. of Trees [2]
Abies balsamea (L.) Mill.	Balsam fir	AFP	51,656
Acer rubrum L.	Red maple	AFP	50,571
Acer saccharum L.	Sugar maple	AFP	33,998
Betula alleghaniensis Britton	Yellow birch	AFP	9,822
F. grandifolia Ehrh.	American beech	AFP	49,246
Picea rubens Sarg.	Red spruce	AFP	23,403
Pinus strobus L.	Eastern white pine	AFP	3,380
Prunus serotina Ehrh.	Black cherry	AFP	16,393
Tsuga canadensis (L.) Carriere	Eastern hemlock	AFP	12,701

Table 1. *Cont.*

Species	Common name	Region [1]	No. of Trees [2]
Picea glauca (Moench) Voss	White spruce	AK	415
Abies lasiocarpa (Hook.) Nutt.	Subalpine fir	IM	1,141
Picea engelmannii Parry ex Engelm.	Engelmann spruce	IM	933
Nothofagus solandri (Hook. f.) Oerst. var. *cliffortioides* (Hook. f.) Poole	Mountain beech	NZ	8,645
Abies amabilis Douglas ex J. Forbes	Pacific silver fir	PNW	5,569
Taxus brevifolia Nutt.	Pacific yew	PNW	1,057
Thuja plicata Donn ex D. Don	Western red cedar	PNW	1,076
Tsuga heterophylla (Raff.) Sarg.	Western hemlock	PNW	7,502
Pinus ponderosa Douglas ex C. Lawson	Ponderosa pine	PNW & IM	5,604
Abies concolor (Gord.) Lindl. Ex Hildebr.	White fir	SN	6,230
Abies magnifica A. Murray	Red fir	SN	3,642
Calocedrus decurrens (Torr.) Florin	Incense cedar	SN	2,383
P. lambertiana Douglas	Sugar pine	SN	1,495

[1]: AFP=Adirondack Forest Preserve, NY; PNW=Pacific Northwest Region, SN=Sierra Nevada Region, AK=Alaska, IM=Intermountain Region, NZ= New Zealand; [2]: AFP plot data originate from expanded counts on 462 prism plots, AK data from one 20 ha stand, IM data from 26.8 ha total sampling area with a mean plot size of 3 ha, NZ data from 4.3 ha total sampling area from 107 plots of 0.04 ha each, PNW data from 51 ha total sampling area (mean plot size 1.1 ha), PNW and IM data from 77.8 ha total sampling area (mean plot size 1.4 ha), and the SN data from 23.2 ha total sampling area (mean plot size 1.2 ha).

The datasets for the 11 conifer species in old-growth stands of the Intermountain (IM), Pacific Northwest (PNW), and Sierra Nevada (SN) regions of the western US included growth and mortality measurements for trees monitored for 25–72 years dependent upon region [14]; but we assessed the structural sustainability of those 11 species using their most recent survey data (1998–2007). The Adirondack Forest Preserve (AFP) dataset consisted of data from nine species located on 462 randomly established prism plots mostly on New York state land outside the AFP, but also within the 1,000,000 ha AFP in New York. The AK dataset consisted of white spruce trees from a 20 ha stand near Anchorage, AK under threat of an outbreak of spruce beetle (*Dendroctonus rufipennis* Kirby). We recognize that this dataset was obtained from a single stand. We utilized it because the stand was uneven-aged, contained a sufficient number of trees for analysis, all diameter classes were present, and the stand reflected the site and management history of the surrounding forest. The New Zealand

dataset was comprised of Mountain beech trees on 107 plots located within a mosaic of even-aged stands on the Harper Watershed located on the South Island.

Each dataset included the number and diameter at breast height (dbh) of every live and dead tree of the species of interest on the sampled plots. The total sampled area for each species was greater than 1 ha because complete and reliable negative exponential diameter distributions cannot be generated in most cases from a smaller sampling area [10]. Average growth rates for each species either were determined directly (e.g., old-growth western conifer datasets) or were obtained from silvics manuals, or regional online databases. The DC bin size for both living and dead trees was determined by the period of time that dead trees of a given species remained identifiable in the field. This time period was estimated through personal experience or discussion with landowners, foresters, or others familiar with the species in the specific region. Then, DC bin size is determined by multiplying the mean growth rate (cm/yr) by the decay rate (yr). This approach assures that both baseline mortality and observed mortality figures are comparable. Identification of dead wood to species may be easier for some species than others, and could prove problematic for some. Therefore, some species that decay very rapidly or very slowly upon death may not be amenable to BM analysis.

2.2. Methodological Assumptions

The baseline mortality method is based upon certain assumptions [3]: (1) baseline mortality is a landscape not a stand-level analytical method. Indeed, stand-level factors are ignored in these analyses; (2) appropriate datasets for analysis must accurately reflect the diameter distribution present. All of the species populations that we utilized in these analyses had a density >20 stems/ha. (3) sampling plots ideally should be randomly located to minimize potential biases for representing the population; (4) all dead trees, including standing, downed, leaning, and stumps, must be counted, regardless of the cause of mortality; and (5) the results of baseline mortality analysis must be interpreted in the context of the ecological and life-history characteristics of the species.

2.3. Baseline and Observed Mortality

The baseline mortality of each species population was calculated from the diameter distribution of the living trees using previously described methods [4,9] as follows: If the diameter distribution of a species is approximately a reversed-J shape, this negative exponential relationship between living tree density (N) and the middle value of the diameter class (D) can be regressed by a log-linear model using ordinary least squares (OLS):

$$\ln(N) = \alpha_0 + \alpha_1 \times D \tag{1}$$

where ln is natural logarithm, α_0 and α_1 are regression coefficients to be estimated from the data. If the R^2 of the log-linear model is 0.80 or greater, the resulting model is considered to satisfactorily fit the diameter distribution of that species population.

Then, the baseline mortality (BM) rate is calculated as follows:

$$BM = 1 - e^{-\alpha 1 \times \Delta D} \tag{2}$$

where α_1 is the estimated slope coefficient of Equation (1), and ΔD is the DC bin size for the species. Expected mortality can be calculated for each DC by multiplying the number of trees in each DC by BM, which represents the desirable mortality in a structurally stable forest. The difference between the observed and expected mortality indicates the disparity of structural sustainability for a given species. Whether or not the difference is statistically significant in one or more DC classes can be determined by a chi-square test using the observed and expected mortality for the species distribution, as well as for each diameter class. When the expected mortality is less than five trees in a DC, adjacent DCs can be combined to satisfy the assumptions of chi-square test.

Observed mortality is determined by counting all dead trees on the site regardless of the cause of the mortality and when it occurred. This mortality includes standing, leaning, and downed dead trees, as well as stumps. The diameter of all the dead trees and stumps is recorded. This number then is divided by the total number of trees in that diameter class and multiplied by 100 to generate percent observed mortality values by diameter class.

2.4. Impacts of Current Mortality on Future Structure

We believe it is essential to provide a forward-looking dimension to the mortality analyses described below. Therefore, we utilized an iterative procedure to evaluate the potential impacts of the currently observed mortality on the future diameter structure, if the current mortality were to remain constant. We assumed that in-growth was constant, *i.e.*, trees growing into the smallest DC were equal to the number of living trees in the smallest DC. For each DC previously shown to have a significant difference between the observed and expected mortality, the observed mortality was used to compute the predicted number of dead trees in that class. Otherwise, the baseline mortality was applied to those DCs where the differences between the observed and expected mortality were not significant. To simulate recruitment, the subsequent iteration began by moving the number of surviving trees in each DC of the last iteration into the next larger DC. Equation (1) was used to predict the number of trees that was added to the now vacant smallest DC in order to simulate constant in-growth. Respective mortality levels defined above were then applied and the number of surviving trees was calculated for each DC. The observed and expected mortality were held constant for each DC, not for respective cohorts of trees. Iterations were continued until the distribution stabilized, and the number of iterations for a given diameter distribution was equal to the number of DCs. Control distributions were generated by applying the baseline mortality to each DC value generated by the OLS model. This is a conservative modeling approach because we recognize that mortality, ingrowth, and recruitment are not likely to remain constant. In addition, the intention is not to model or to predict actual future diameter distributions, for which transition matrix modeling techniques [15] are more suitable.

2.5. Assessment of Structural Sustainability

The structural sustainability of these 22 species was assessed using multivariate statistical procedures based on the following five metrics derived from the patterns of observed mortality in the diameter distributions, and their effects on future diameter distributions as discussed in the above section.

(1) Distribution of mortality (DM) is the distribution of DCs in which the observed mortality significantly differed from the expected mortality. DM is coded as 5 if the significant difference is in <25 cm DCs, 4 if in 25–38 cm DCs, 3 if in 38–64 cm DCs, or 2 if in >64 cm DC. These DM codes are based loosely on pole, small, medium, and large sawtimber size classes in the northern hardwood forest. It reflects the assumption that excessive or insufficient mortality in the smaller size classes disproportionately impacts the future diameter distributions relative to the larger size classes.

(2) Aggregation (AGG) is the clustering of significantly different DCs. AGG is coded as (1) if there are 2–3 consecutive DCs with significant differences between the observed and expected mortality, (2) if 4–5 consecutive DCs, (3) if 6–7 consecutive DCs, or (4) if >8 consecutive DCs. It reflects the greater effect of mortality clustered in adjacent DCs compared to non-aggregated mortality. Distributions without consecutively different DCs were assigned a value of zero.

(3) Magnitude (MAG) is the size of the differences between the observed and expected mortality for the species. MAG is the summation of the absolute values of the differences between the observed and expected mortality for each DC.

(4) Relative abundance (RA) is the percentage of DCs that showed significant differences between the observed and expected mortality. It is scored by dividing the number of DCs with significant differences by the total number of DCs in the species distribution.

(5) Change (CHG) is how the control and predicted diameter distributions change from the first to the last iteration. It is evaluated by using chi-square analysis to compare the distribution of predicted living trees (based on the observed and expected mortality—see above) to the control distribution (based on the baseline mortality). The total number of DCs with significant differences between the observed and expected mortality is tallied for each distribution and the difference divided by the total number of DCs, which was then arcsine transformed. This variable was created as stated previously to provide a forward-looking dimension to the mortality analyses.

Based on the five metrics above, hierarchical cluster analysis with the average linkage clustering method was conducted. The 22 species were clearly divided into two clusters. Then, a bootstrapped discriminant analysis using the same five metrics with 1000 iterations was conducted to generate a discriminant function (DF) as follows [16]:

$$DF = \gamma_1 \times DM + \gamma_2 \times AGG + \gamma_3 \times MAG + \gamma_4 \times RA + \gamma_5 \times CHG \tag{3}$$

where $\gamma_1 - \gamma_5$ are discriminant function coefficients estimated from the data. Given the values of the five metrics, the score of the DF equation (Equation (3) was computed for each species, which was considered the "Sustainability Score").

Further, given the fact that these 22 species were clustered into two distinguished groups (coded as S (sustainable) and U (unsustainable)), a logistic model was used to regress the probability of a species i belonging to the group S, *i.e.*, $P_i = P_r (Species_i = S)$, against its "Sustainability Score" as follows:

$$P_r(Species_i = S) = \frac{1}{1 + e^{-(\beta_0 + \beta_1 \cdot Score_i)}} \qquad (4)$$

where β_0 and β_1 are the regression coefficients to be estimated from the data. Thus, the natural log-transformation of the odds ($\frac{P_i}{1-P_i}$) is expressed as a linear function of the "Score":

$$\ln\left(\frac{P_i}{1-P_i}\right) = \beta_0 + \beta_1 \cdot Score_i \qquad (5)$$

If we assume that the probability of species belonging to the sustainable or unsustainable group is 0.5 (equal chance), *i.e.*, $P_i = P_r (Species_i = S) = 0.5$, we can solve for a "sustainability threshold" by letting.

$$\ln\left(\frac{P_i}{1-P_i}\right) = \ln\left(\frac{0.5}{1-0.5}\right) = \ln(1) = 0 = \beta_0 + \beta_1 \cdot Score_i \qquad (6)$$

Therefore, the sustainability threshold is:

$$Threshold = \frac{-\beta_0}{\beta_1} \qquad (7)$$

If the sustainability score of a species calculated using Equation (3) is smaller than the "threshold", the species belongs to the sustainable group, otherwise, it belongs to the unsustainable group.

3. Results and Discussion

All of the species that we evaluated showed negative exponential diameter distributions with model $R^2 > 0.82$. Therefore, baseline mortality (BM%) was constant across all diameter classes for each of the 22 species (Table 2).

Table 2. Baseline percent mortality, correlation coefficient of regression of natural log of diameter distribution fit to the negative exponential model, diameter distribution metrics, discriminant function score, and cluster category for each of the 22 species populations.

Species	BM (%) [1]	R^2 [2]	MAG [3]	AGG [4]	DM [5]	CHG [6]	RA [7]	DF Score [8]	Cluster [9]
A. amabilis	18.1	0.98	4.7	0	5	0.70	0.036	4.3	S
T. plicata	11.3	0.86	19.4	1	5	0.70	0.071	12.9	S
T. brevifolia	30.2	0.88	29	0	5	0.52	0.167	17.6	S
A. magnifica	11.3	0.89	31.3	1	5	0.61	0.075	19.4	S
T. heterophylla	18.1	0.96	38.9	2	12	0.61	0.179	27.2	S
P. glauca	31.6	0.96	46.0	1	5	0.34	0.222	27.6	S
A. concolor	13.9	0.92	65.8	2	7	0.67	0.172	39.6	S

Table 2. *Cont.*

Species	BM (%) [1]	R [2]	MAG [3]	AGG [4]	DM [5]	CHG [6]	RA [7]	DF Score [8]	Cluster [9]
A. lasiocarpa	23.7	0.82	91.6	2	5	0.44	0.357	52.9	S
A. saccharum	20.5	0.96	89.1	2	9	0.69	0.267	53.0	S
B. alleghaniensis	18.1	0.95	89.9	2	12	0.58	0.226	54.7	S
T. canadensis	20.5	0.95	98.1	2	5	0.54	0.31	56.3	S
P. strobus *	13.9	0.82	144.4	2	12	0.63	0.296	84.1	S
P. engelmannii *	21.3	0.91	107.3	3	9	0.38	0.375	63.8	U, +
P. rubens *	25.9	0.95	110.5	3	9	0.48	0.333	65.4	U, −
N. solandri	33.0	0.86	144.7	3	12	0.48	0.54	85.2	U, −
C. decurrens	13.9	0.89	175.7	5	14	0.30	0.351	104.2	U, −
P. serotina	21.3	0.92	192.2	5	12	0.37	0.56	112.3	U, +, −
A. balsamea	40.5	0.98	228.1	4	9	0.42	0.60	129.7	U, −
A. rubrum	26.7	0.95	224.5	6	12	0.28	0.64	130.5	U, −
P. ponderosa	38.1	0.96	246.7	4	14	0.19	0.56	141.9	U, −
F. grandifolia	30.2	0.96	334.9	6	12	0.39	0.71	190.0	U, +, −
P. lambertiana	14.8	0.82	438.0	7	14	0.28	0.72	247.2	U, +

[1] baseline percent mortality; [2] correlation coefficient of regression of natural log of diameter distribution fit to the negative exponential model; [3] magnitude metric value; [4] aggregation metric value; [5] distribution metric value; [6] change metric value; [7] relative abundance metric value; [8] discriminant function score=sustainability score; [9] S: sustainable, U: unsustainable, +: excessive mortality, −: insufficient mortality; * misclassified by discriminant function analysis.

Cluster analysis using the five mortality metrics classified the 22 species populations into two clusters. The discriminant analysis on the two clusters yielded one significant canonical root (Wilk's lambda = 0.2194; F (5,16) = 11.38; $p < 0.0001$], confirming the two clusters were statistically distinguishable. Moreover, all five metrics were significantly ($p < 0.05$) important for cluster separation. The following discriminant function based on the 1000 bootstrapped discriminant analyses was derived from the mean pooled within-class canonical coefficients:

$$DF = (0.699) \times AGG + (0.684) \times RA + (0.535) \times MAG + (0.420) \times DM − (0.554) \times CHG \qquad (8)$$

then, the score of the *DF* function (Equation 8) was computed for each species (Table 2).
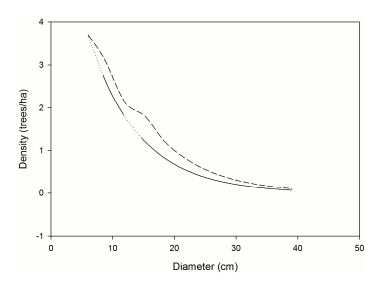
We view the bootstrapped discriminant function as a "structural sustainability index" because the five metrics were based on patterns of observed mortality and existing diameter distributions, and were designed to compare mortality among different populations of trees, and to reflect potential future changes in the size structures of the species populations. For the sustainable group, the lowest sustainable score was 4.27 and the highest score was 56.3 (mean = 33.2 and median = 27.6). For the unsustainable group, the lowest unsustainable score was 85.2 and the highest score was 247.2 (mean = 142.6 and median = 130.1). The threshold score (Equation 7), defined as the sustainability score with an equal chance of being designated sustainable or unsustainable, was derived from logistic regression model parameters in Equation (4) (*i.e.*, $\beta_0/−\beta_1 = −7.1002/0.1006 = 70.58$) Thus, species with scores less than 70.58 were sustainable and those with scores over this threshold were unsustainable.

Discriminant analysis had an apparent total error rate of 0.14 (0.08 for sustainable group and 0.20 for unsustainable group). *P. engelmannii* and *P. rubens* were misclassified as sustainable, and *P. strobus* was misclassified as unsustainable (Table 2).

Sustainability problems were evident in each temperate forest type, and in each of the three old-growth western coniferous forest regions (PNW, IM, and SN), although most of the unsustainable species were located in the Adirondack Forest Preserve of New York (Tables 1 and 2). The results of individual baseline mortality analyses, however, apply only to the species in the forests that were assessed, and should not be extrapolated to represent the sustainability of that species in other forested regions.

In total, 12 of the 22 species populations were categorized as structurally sustainable and 10 as unsustainable (Table 2). Significant deviations between observed and baseline mortality in some DCs does not necessarily threaten the structural sustainability of a species. For example, the sustainable diameter distribution of *T. brevifolia* (PNW, Figure 2) illustrates overall congruence of the projected and control diameter distributions even though significant differences between observed and expected dead occur in some diameter classes.
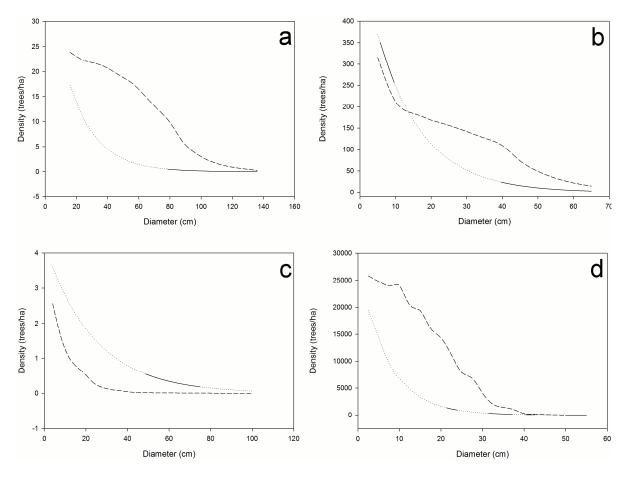
Figure 2. Calculated distributions illustrating the impact of currently observed mortality on sustainability of *Taxus brevifolia* (PNW); congruence of control (—) and projected (- - -) size class structures indicates that the observed mortalities will not significantly impact forest structure; diameter classes with significant differences between observed and baseline mortality are shown as dotted lines; dashed line segments above or below the solid/dotted line indicate insufficient or excessive mortality, respectively; future diameter distributions hold ingrowth, recruitment, and mortality constant.

Structural unsustainability can be caused by either excessive or insufficient mortality (Figure 3a–d). Indeed, current mortality levels of ponderosa pine (Figure 3a) in the sampled old-growth stands in the PNW and IM regions are insufficient (*i.e.*, observed significantly less than expected dead) to maintain their structural stability. The accumulation of *P. ponderosa* is attributable to decades of fire suppression [17,18], which may explain the low mortality that we observed (Figure 3a). High density stands of low vigor often result in outbreaks of mountain pine beetle (*Dendroctonus ponderosae*,

MPB) and catastrophic crown fires, which have occurred in pine stands in both Arizona and Oregon [18,19]. Thus, the baseline mortality method successfully identified this well-recognized problem in old-growth ponderosa pine forests of the western United States. One potential problem using this method in such high-severity, fire-prone ecosystems is that small trees might be completely consumed by fire that may not allow all dead trees across all diameter classes to remain identifiable. In such cases, a lower diameter limit could be identified for baseline mortality determination.

Figure 3. Calculated distributions illustrating the impact of currently observed mortality on sustainability of (**a**) *Pinus ponderosa* (PNW + IM), (**b**) *Nothofagus solandri* (NZ), (**c**) *Pinus lambertiana* (SN), and (**d**) *Fagus grandifolia* (AFP); lack of congruence of control (—) and projected (- - -) size class structures indicates that the observed mortalities will significantly impact forest structure; diameter classes with significant differences between observed and expected dead trees are shown as dotted lines; dashed line segments above or below the solid/dotted line indicate insufficient or excessive mortality, respectively; future diameter distributions hold ingrowth, recruitment, and mortality constant.

Nothofagus solandri exists as a mosaic of even-aged stands on both the North and South Islands of New Zealand. Stand structure in these forests is influenced by large-scale disturbances, such as the snowstorms of 1968 and 1973, and the 1994 earthquake; which caused widespread landslides and subsequent tree damage and mortality on the South Island [20]. The response to these events created an unsustainable structure consisting of too many trees (insufficient mortality) in the small to intermediate

size classes (diameter classes 2–10 in Figure 3b represent diameters of 10–40 cm, respectively) from the 1999 dataset we evaluated (Table 2, Figure 3b). Hurst *et al.* [21] observed annual mortality rates in these same stands to decrease from 0.022 in 1974–1983 to 0.019 in 1983–1993, and finally to 0.018 in 1993–2004; and that this decrease was not attributable to crowding effects over time. Our analysis confirms their conclusions, and suggests that future, excessive mortality is likely, and will be necessary to restore landscape-level sustainability to these forests.

Excessive mortality (*i.e.*, observed significantly greater than expected dead) also can cause structural sustainability problems (Figure 3c,d). High levels of mortality across all diameter classes of *P. lambertiana* in the SN region (Figure 3c) is most likely attributable to white pine blister rust [14]; a non-native, invasive fungal disease of five-needle pines in North America [22] that could extirpate this species from these stands should it continue at these high levels. The baseline mortality method confirmed excessive levels of mortality, which we attribute to this devastating invasive pathogen.

In the northern hardwood forests of the AFP, our analyses identify five structurally unsustainable species populations including *A. balsamea, A. rubrum, F. grandifolia, Picea rubens,* and *P. serotina* (Table 2). The unsustainability of American beech is attributable to the invasive disease complex known as beech bark disease, which explains the insufficient mortality in the small DCs and excessive mortality in the larger DCs (Figure 3d) [23]. Beech bark disease induced-mortality of large, mature trees results in dense thickets of young beech of root-sprout origin. The current and projected structures are undesirable from a management perspective because the excess of small stems and death of larger trees are significant management challenges. Nonetheless, knowing the current size structure and how it is changing provides an opportunity to develop proactive management strategies for beech. Although unknown, we suspect that the cause of unsustainability of *P. serotina* may be attributable to past cutting practices in this region. The insufficient mortality observed in *A. balsamea* and *P. rubens* may be a natural response to a regionwide windstorm in 1995 that caused severe blowdown. Similarly, insufficient mortality observed in the *A. rubrum* population may be the result of a systematic successional shift in the abundance of this species that is occurring in many areas. All structural changes must be interpreted in the light of the ecological characteristics of the species, disturbance history, and whether or not the changing structure interferes with the management objectives for the forest. Periodic BM analyses for these species could reveal whether the structural unsustainability was temporary or becomes a forest health issue.

Increasing rates of mortality detected in the sampled old-growth coniferous forest stands of the western U.S. was attributed to climate change by van Mantgem *et al.* [14]. However, their study utilized pooled species data so extrapolation of their conclusions to our individual species analyses should be made with caution. Nevertheless, the baseline mortality method suggests that the increasing mortality they observed is excessive for *P. lambertiana*, the cause of which is white pine blister rust; and *P. engelmannii* for which the cause is unknown. The other old-growth stands of western conifer species that were structurally unsustainable are *C. decurrens* in the SN region, and *P. ponderosa* in the PNW and IM regions, which are both attributable to insufficient mortality.

Structural stability (*i.e.*, sustainability, assessed this way) has several advantages over other methods. First, dead trees are comparatively easy to recognize and to count, whereas the ecological processes that regulate a forest ecosystem (e.g., energy and nutrient flow, trophic-level interactions, ecosystem balance, and resilience) are not. Second, forest health monitoring programs typically focus

on tree crown condition over time to gauge the health of a forest [24]. However, crown condition is reversible, and its measurement subjective. More importantly, reliance on crown health confuses individual tree health (organism level) with forest health (population to landscape level). Third, the baseline mortality approach provides a framework within which to detect elevated (or insufficient) levels of mortality. Structurally sustainable, healthy forests have dead and dying trees, but now we can determine if that mortality is too much or too little. Fourth, routinely collected forest inventory data (species, growth rates, number and size of living and dead trees) are sufficient to conduct these analyses. Lastly, baseline mortality and its relationship to observed mortality can provide a forward-looking, prospective approach to the assessment of structural sustainability. Tree mortality data are collected annually by the USDA Forest Service [25] and agencies in other countries and trends in mortality often are evaluated retrospectively by comparing current to historical mortality levels. This approach assumes that historical mortality levels are an appropriate frame of reference, which may not be true. Baseline mortality provides a more rational baseline to which currently observed mortality can be compared, and has the added advantage of being able to prospectively assess the impacts of current mortality levels on future forest structure and composition.

The multivariate structural sustainability index and threshold value are empirically derived from 21 temperate and one boreal forest tree species. It may eventually be possible to develop a global sustainability index once more forest species from the temperate, boreal, tropical and subtropical forests of the world are evaluated. At that time it may be possible to compare the impact of global disturbances (e.g., climate change) on forest structural sustainability both spatially and temporally. Until then, we propose that a regionally derived structural sustainability index score be included as one more additional and important consideration when forestland managers or landowners assess the sustainability/health of their forests.

Although the baseline mortality concept and associated index may not be appropriate for certain species (e.g., those with diameter distributions other than negative exponential), the ramifications of our proposed methods are significant. A standardized analysis that permits the objective and quantitative assessment of the structural sustainability of a forest provides the ability to determine the impact of natural and anthropogenic disturbance agents, including management activity, on that forest. We have used the method to verify the impacts of known disturbance agents on four species, and the results support our hypothesis that the method can identify or elucidate structurally unsustainable forests. Plus, it provides the ability to recognize a potential sustainability problem before the cause is known, *e.g.*, *A. balsamea, A. rubrum, C. decurrens, Picea engelmannii, P. rubens*, and *P. serotina* (Table 2); and to alert forest health professionals and agencies to the potential need for action. This approach may enable comparison of the impacts of disturbances in time and space on forests on a global scale. Government agencies, NGOs and forest health professionals also can prioritize resource allocation and research needs based upon an objective assessment of how native and invasive pests and pathogens as well as other disturbances affect structural sustainability. Sturrock *et al.* [26] call for a triage system to help ecologists and forest managers decide which forest types or species are most threatened by climate change or other disturbances and to develop long-term management plans. Finally, distinguishing between stands with different structural conditions is an essential feature of all useful structural indices that can reveal clues to the ecological processes that drive structure [27]. Methods that can differentially distinguish stands with changing structures are needed [27]. The

baseline mortality approach answers these calls, and promotes the objective assessment of forest structural sustainability in a manner that is unique in its clarity and globally applicable.

A software package is available at the following URL [28] that performs all of the calculations for these analyses once all of the necessary data (dbh and health status (living or dead) of all trees of the species of interest in the area, and DC bin size) are input.

4. Conclusions

Structural sustainability is a key criterion to an objective and quantitative assessment of forest health. Forest structure is an ecologically and demographically sound and comparatively easily measured feature of all forests using readily available inventory data upon which to base a method for assessing sustainability. Here, we present a quantitative index of structural sustainability that objectively distinguishes sustainable from unsustainable diameter distributions, and propose it as one additional but critical criterion for the assessment of forest health.

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Author Contributions

Jonathan A. Cale, Justin L. West, and David R. Castello conducted all of the baseline mortality analyses in this study. David R. Castello and Peter Devlin developed the computer software package to facilitate the use of the method by others. Jonathan A. Cale, John D. Castello and Stephen A. Teale developed the quantitative baseline mortality method itself. Lianjun I. Zhang provided all of the statistical expertise. The concept of a structural sustainability index was conceived by John D. Castello and Jonathan A. Cale.

Conflicts of Interest

The authors declare no conflicts of interest.

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