

Modeling the Effects of Fire Frequency and Severity on Forests in the Northwestern United States

Scientific Investigations Report 2006–5061

**U.S. Department of the Interior
U.S. Geological Survey**

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By Richard T. Busing and Allen M. Solomon

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**U.S. Department of the Interior
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Suggested citation:

Busing, Richard, Solomon, Allen, 2006, Modeling the Effects of Fire Frequency and Severity on Forests in the Northwestern United States: U.S. Geological Survey Scientific Investigations Report 2006-5061, 16 p.

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Conversion Factors, Datums, Definitions, and Abbreviations and Acronyms

Conversion Factors

Multiply	By	To obtain
Length		
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
mile, nautical (nmi)	1.852	kilometer (km)
yard (yd)	0.9144	meter (m)
Area		
acre	4,047	square meter (m ²)
acre	0.4047	hectare (ha)
acre	0.4047	square hectometer (hm ²)
acre	0.004047	square kilometer (km ²)
square foot (ft ²)	929.0	square centimeter (cm ²)
square foot (ft ²)	0.09290	square meter (m ²)
square inch (in ²)	6.452	square centimeter (cm ²)
section (640 acres or 1 square mile)	259.0	square hectometer (hm ²)
square mile (mi ²)	259.0	hectare (ha)
square mile (mi ²)	2.590	square kilometer (km ²)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$^{\circ}\text{F}=(1.8\times^{\circ}\text{C})+32$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$^{\circ}\text{C}=(^{\circ}\text{F}-32)/1.8$$

DBH is the abbreviation for stem diameter at breast height (1.37 m above ground)

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Altitude, as used in this report, refers to distance above the vertical datum.

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Abstract

This study used a model of forest dynamics (FORCLIM) and actual forest survey data to demonstrate the effects of various fire regimes on different forest types in the Pacific Northwest. We examined forests in eight ecoregions ranging from wet coastal forests dominated by *Pseudotsuga menziesii* and other tall conifers to dry interior forests dominated by *Pinus ponderosa*. Fire effects simulated as elevated mortality of trees based on their species and size did alter forest structure and species composition. Low frequency fires characteristic of wetter forests (return interval >200 yr) had minor effects on composition. When fires were severe, they tended to reduce total basal area with little regard to species differences. High frequency fires characteristic of drier forests (return interval <30 yr) had major effects on species composition and on total basal area. Typically, they caused substantial reductions in total basal area and shifts in dominance toward highly fire tolerant species. With the addition of fire, simulated basal areas averaged across ecoregions were reduced to levels approximating observed basal areas.

Introduction

The ecological impacts of severe forest fires are significant. Loss of live forest biomass and sharp declines in the abundance of most tree species are well-known consequences of severe fire events (Whelan, 1995). The effects of less severe fire events on forests are more subtle, but the cumulative effects of multiple low-severity fires demand attention. Across the Pacific Northwest, forests are subject to a wide diversity of fire regimes (Agee, 1993). It follows that the region is well-suited for investigation of contrasting fire regimes on forest dynamics.

How fire severity and frequency impact forest composition and structure is best studied or simulated over the long periods of time required to generate forest tree life cycles. This research requires mechanistic models that consider both tree species and stem size susceptibility to fire. Models that can simulate changes in forest dynamics over centuries are also needed to investigate the effects of global climate change and shifting fire regimes on forests. The FORCLIM forest gap model (Bugmann, 1996) is suitable for such investigations. It operates on time scales from individual years to centuries and

it simulates changes in both species composition and the size of tree stems. It also projects the effects of climate on trees (Bugmann and Solomon, 2000), so it is a versatile tool in the study of forest responses to environmental change. With the added capability of simulating fire frequency and severity it becomes particularly useful for simulating forest dynamics in the western U.S. Our objectives are to: 1) develop a fire disturbance routine for FORCLIM, and 2) apply the model in evaluating the consequences of fire frequency and severity on a variety of forest types representative of forests in the western U.S.

This study is novel because other forest gap models that include wildfire routines do not generate realistic results across the broad gradients or variable terrain of the Pacific Slope regions of the western U.S. (e.g. ZELIG, Miller and Urban, 1999; see Busing and Solomon, 2004). None of the gap models applied to these regions (e.g. CLIMACS, Dale and Hemstrom, 1984; SILVA, Kercher and Axelrod, 1984; FIRE-SUM, Keane et al., 1990; ZELIG, Urban et al. 1993) have a demonstrated ability to simulate vegetation forcing by and response to seasonal moisture variations, which control both species composition and fire regimes in the Pacific Northwest.

Purpose and scope

In earlier modeling investigations we demonstrated that FORCLIM was able to simulate forest composition and structure across broad climatic gradients in western Oregon (Busing and Solomon, 2004, 2005). This was particularly true in forests subject to infrequent fire. However, in some ecoregions with high fire frequencies, simulated composition and structure did not match that of actual forests.

In this study we examine whether the addition of fire disturbance to the simulations improves agreement between simulated and actual forest characteristics. We simulate a wide range of fire regimes from high severity, low frequency events to low severity, high frequency events to see how fire regime affects forest structure and composition. By way of simulation we also contrast these regimes to those devoid of fire disturbance. Comparison of various simulated forests to actual forests is made. In this way we test the suitability of the FORCLIM model in forests of the northwestern United States. We also gain understanding of the long-term effects of fire on forest dynamics.

The study region

To evaluate the modeled fire routine and its simulated fire effects we chose a series of eight ecoregions along a broad climatic gradient in western Oregon from the Pacific Coast to the interior high desert (Fig. 1). The study region spanned a wide variety of climates, forest types and fire regimes representing many forested regions of the western United States. Coastal forests of the region are some of the wettest forests in the western United States with mean annual precipitation exceeding 200 cm (Fig. 2a; Sturnes, 1960; Franklin and Dyrness, 1988). Fires are very infrequent in wet, coastal forests but when such forests are dry enough to burn, fires are almost invariably stand-replacing events (Agee, 1993). Forests and rangelands of interior Oregon east of the Cascades undergo little annual rainfall and experience particularly dry summers (Fig. 2b). They have high frequencies of ground fires and few stand-replacing crown fires, resembling fire regimes in much of the western U.S. (e.g., Swetnam and Betancourt, 1998; Brown and Wu, 2005).

Methods

The empirical data

In order to assess the existing vegetation of western Oregon a set of data on 2323 forest stands in the region was assembled from USDA and USDI databases (Busing, 2004). All stands were inventoried between 1993 and 2002 using USDA Forest Service conventions adopted for an ecological survey of forested federal lands (Max et al., 1996). In that system, known as the Current Vegetation Survey, plots were established on a square grid at 5.5 km intervals. A five-subplot design covering a 1-ha area was used for tree stratum observations at each plot site. Measurements on live trees included diameter at breast height and, for selected individuals at most sites, tree height and age. We selected sites in which all five subplots were inventoried and tree age data were collected.

Our analyses in processing the data for this modeling investigation were restricted to all live trees >10 cm DBH. We calculated basal area for dominant species and for all species combined. Stand age, or time since the last stand-replacing disturbance, was estimated as the maximum age of a set of overstory trees selected for coring at each plot site. Climate data were assigned to each plot site using the geographic coordinates of the plot and interpolated climate data sets. The climate data consist of monthly mean precipitation and temperature from 1961 to 1990 on a 4-km grid across the conterminous United States (Lugo et al., 1999).

The model

The FORCLIM model of forest dynamics is designed to simulate forest responses to disturbance events and to climatic change in space and time (Bugmann, 1996). Assessments of simulated forest composition along climatic gradients have demonstrated the wide applicability of advanced versions of FORCLIM in temperate forest regions (Bugmann and Solomon, 2000). The model generates tree species composition along climatic gradients that tends to follow patterns of potential natural vegetation (*sensu* Kuchler, 1964). It also simulates stand development and succession as a tree-by-tree replacement process. Projected changes are driven by tree-level responses to precipitation, temperature and disturbance regimes. Thus, the model can simulate forest response to changing regimes of climate and disturbance (Busing and Solomon, 2004, 2005).

The FORCLIM model is individual-based and can operate across a range of spatial and temporal scales (Bugmann and Solomon, 1995). It is a gap model (*sensu* Shugart, 1984) in which a set of independently simulated 0.08-ha plots, each representing a gap-sized patch, is taken to represent a forest stand or landscape (Bugmann, 1996). Compositional and structural dynamics are driven by tree species responses within the dynamic biotic and abiotic environments modeled and assessed annually for each plot. A pool of 18 tree species (20 taxa when two subspecies are distinguished) common to northwestern Oregon is used in this study (Bugmann and Solomon, 2000; Busing and Solomon, 2004; Appendix). Key species parameters affecting dynamics include a maximum diameter at breast height (DBH), a maximum age, a unique set of resource-growth-response functions, and a set of conditions for establishment of new trees. General carrying capacity (or biomass capacity) functions are not used in FORCLIM but rather are emergent properties of the model.

The model applied in this study (FORCLIM version 2.9, Bugmann and Solomon, 2000) resulted from a long series of enhancements to the JABOWA model (Botkin et al., 1972). The fundamental tree competition, growth and mortality considerations in JABOWA were employed in FORCLIM version 1.0 with some modifications (Bugmann, 1994, 1996; Bugmann and Solomon, 1995). They were also used in the southern Appalachian model FORET (Shugart and West, 1977) and the southern central European model FORECE (Kienast, 1987) from which FORCLIM 1.0 evolved. Unique features of FORCLIM 2.9, a version developed for the strongly seasonal climate of the Pacific Northwest, center on the effects of climate on tree performance (Bugmann and Solomon, 2000).

Like earlier gap models (e.g. Botkin et al., 1972; Shugart, 1984), monthly mean temperature and precipitation serve as the basis for climate calculations affecting annual growth of trees. Modifications in FORCLIM 2.9 include a new temperature-growth relationship, a new soil moisture submodel, a deciduous species growing season, and a chilling requirement for establishment. The widely used parabolic curve of potential tree growth versus total annual growing degree days

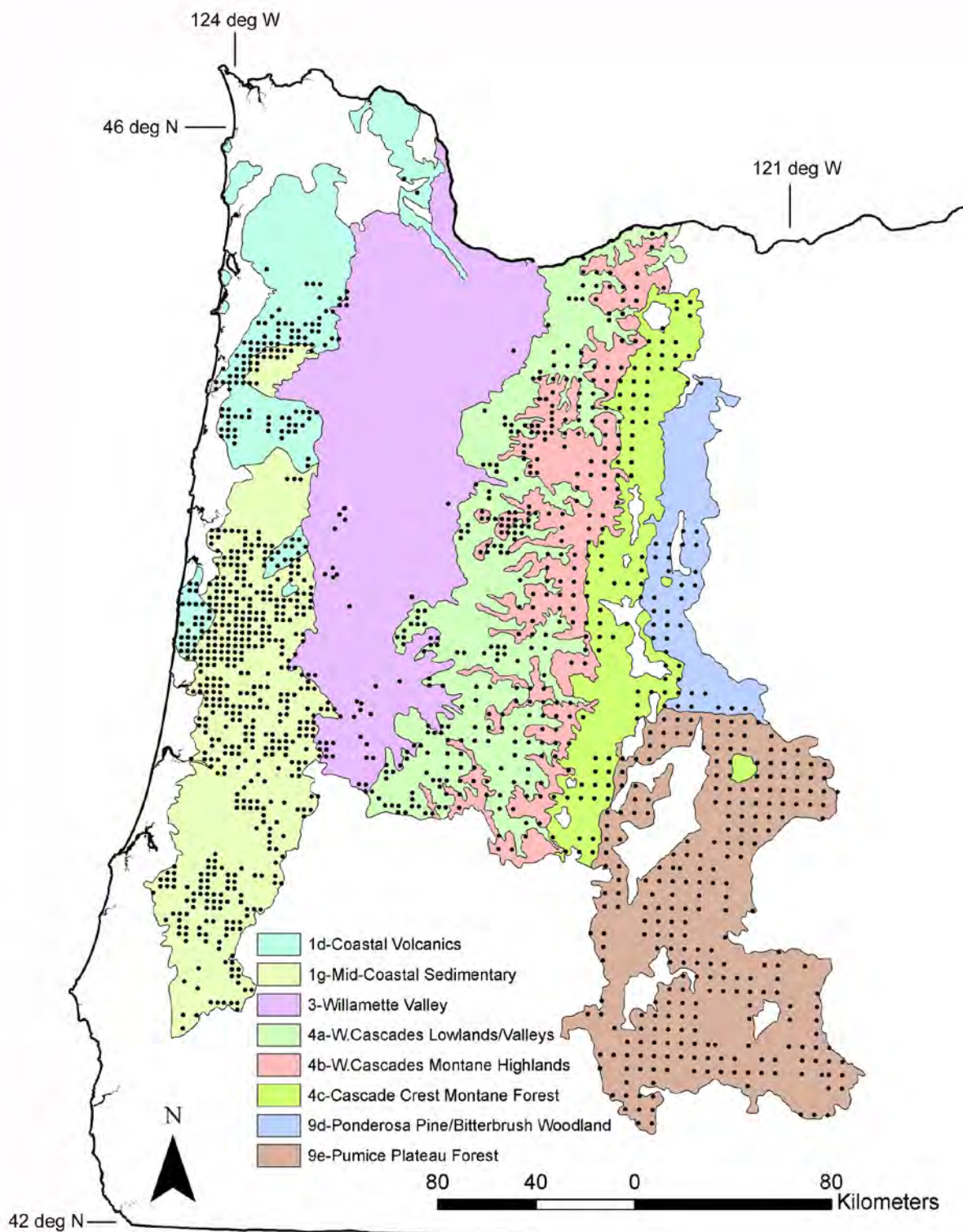


Figure 1. Map of western Oregon showing the ecoregions used in the study (Thorson et al. 2002). The pool of field sites located within each ecoregion is displayed.

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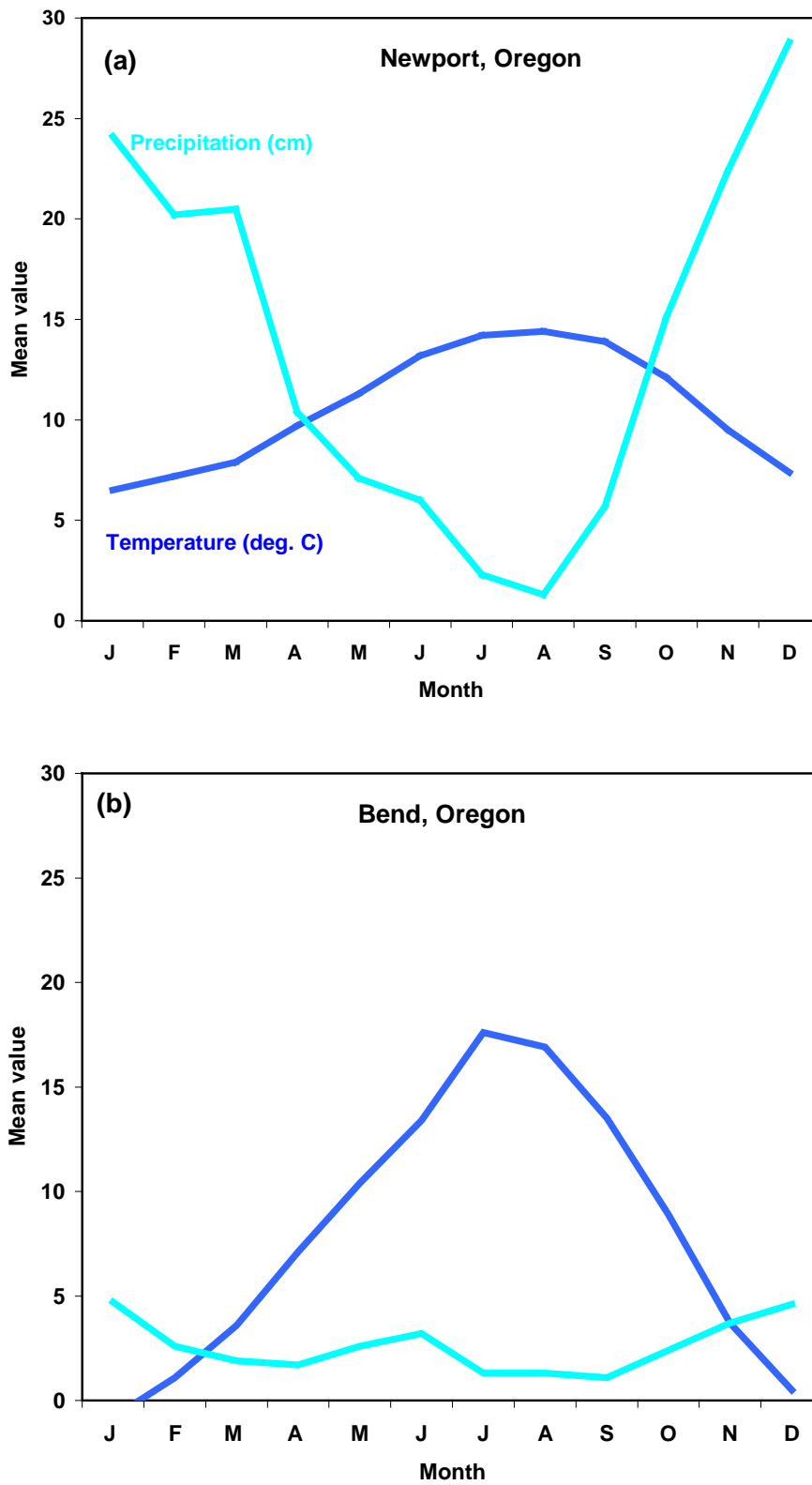


Figure 2. Graphs of monthly climatic means in a) a coastal site (Newport, Oregon), and b) an inland site (Bend, Oregon).

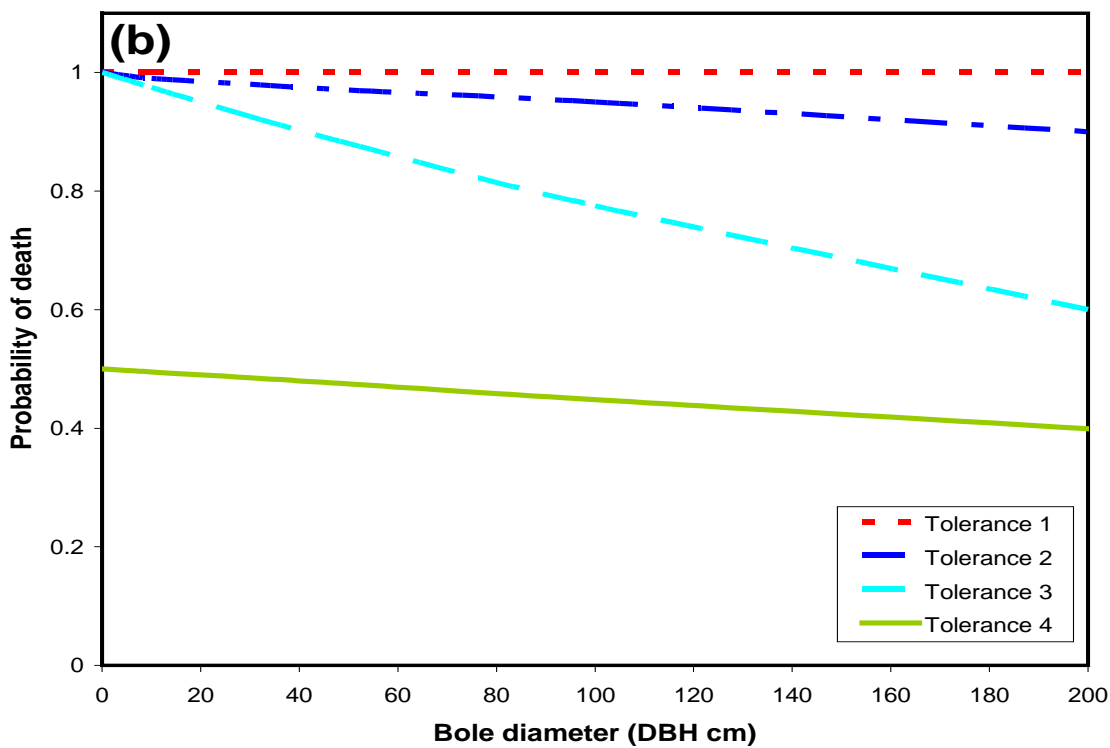
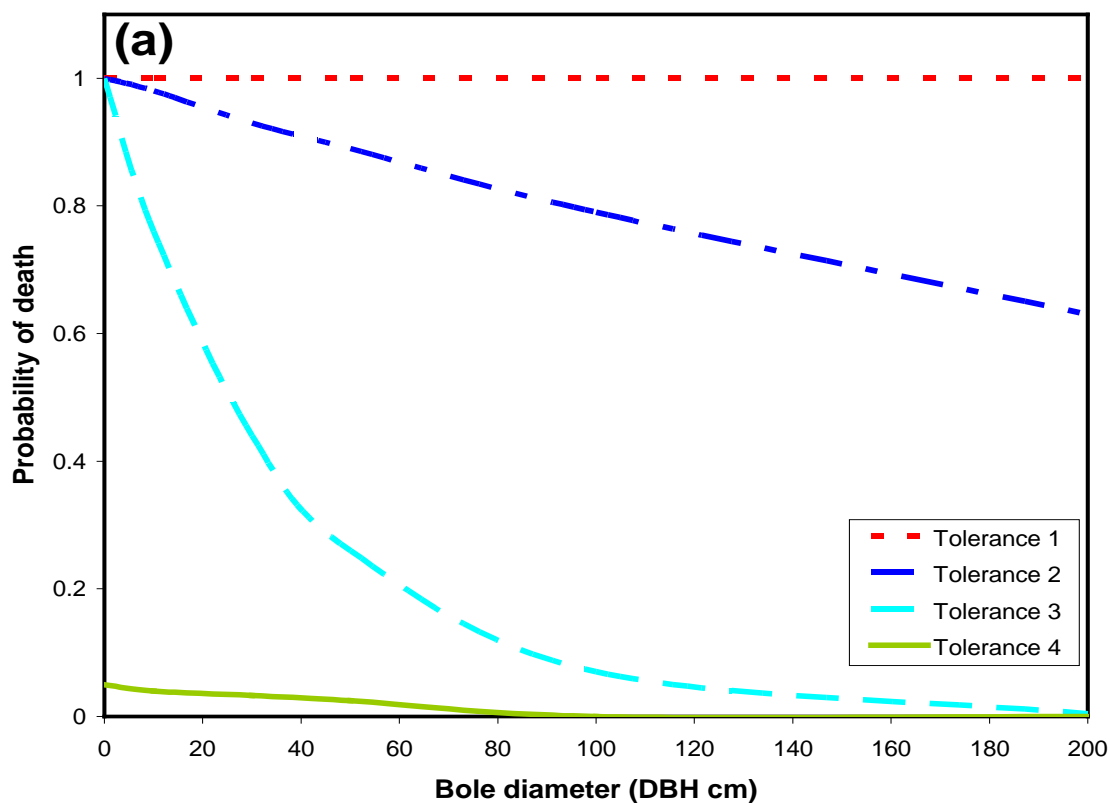


Figure 3. Graphs of mortality responses of trees by stem diameter and fire tolerance class under a) low-severity fire, and b) high-severity fire. Equations are provided in the Appendix.

(Botkin et al., 1972; Shugart, 1984; Botkin, 1992) is replaced with an asymptotic curve assuming reduced potential growth only at the colder edge(s) of a species' range. In addition, the potential growth of deciduous species is determined from climate data only for months when deciduous trees are in leaf (April to October). The winter chilling requirement is added to the regeneration "filters" (sensu Shugart, 1984) for each species. If the simulated minimum winter temperature for a year is too warm to permit winter dormancy, then regeneration of the species is temporarily reduced. This chilling requirement is defined by a species-specific temperature value. A drought index is applied to constrain tree growth based on the species' moisture stress tolerance including their ability to withstand many consecutive weeks with soil moisture below the wilting point. The drought index is associated with a new water balance submodel of soil moisture using a bucket scheme and evapotranspiration estimates (Bugmann and Cramer, 1998) rather than the more commonly used and more empirical submodel derived from Thornthwaite and Mather (1957). These and other modifications leading to version 2.9 of FORCLIM, which is suited to Pacific Northwest forests and climates, are discussed in further detail by Bugmann and Solomon (2000).

We modified the FORCLIM 2.9 model to simulate effects of fire frequency and severity. Species were assigned to one of four fire tolerance classes based on natural history information provided by Minore (1979) and the USDA Fire Effects Database. The least fire tolerant class contained *Abies amabilis*, *Abies lasiocarpa*, *Chamaecyparis nootkatensis*, *Picea engelmannii*, *Picea sitchensis*, *Tsuga heterophylla*, and *Tsuga mertensiana*. These species (class 1) were assumed to undergo complete stand-wide mortality during a fire event regardless of fire severity. For the three remaining classes of tolerance, fire effects were simulated so that large trees had greater survivorship than small trees (Fig. 3; Dale and Hemstrom, 1984). Species in the mildly tolerant category (class 2) (*Abies grandis*, *Abies procera*, *Acer macrophyllum*, *Alnus rubra*, *Arbutus menziesii*, *Pinus contorta*, *Pinus monticola* and *Thuja plicata*) were assumed to suffer heavy mortality, particularly in severe fires. *Pseudotsuga menziesii* was the sole species in the moderately fire tolerant category (class 3). Typically, it suffered low to intermediate levels of mortality. Survivorship depended strongly on stem size and fire severity. Trees in the highly tolerant category (class 4), containing *Pinus ponderosa* and *Quercus garryana*, were subject to relatively low mortality from fire events.

Fires regimes were simulated as a combination of low-severity and high-severity events (see Appendix). Simulated events of low severity had little impact on mortality of larger trees of fire-resistant species. However, trees of fire-intolerant species, and saplings and small trees of species with low to moderate fire tolerance were impacted by events of low severity as well as those of high severity. The likelihood of tree mortality among fire-resistant species (class 2-4) increased with fire severity.

Model comparisons with field data

The ability of FORCLIM 2.9 to generate accurate vegetation patterns across environmental gradients was tested using the forest survey data set. The test involved older forest sites (>150 yr) surveyed within selected ecoregions representing much of the variation in precipitation, temperature, and natural disturbance in forests of western Oregon. Simulations were run for a set of 20 randomly selected sites in each ecoregion having more than 20 sites. The Willamette Valley and Ponderosa Pine/Bitterbrush Woodland ecoregions each had only 20 suitable sites and so all sites from each of these ecoregions were used in the analyses. Critical input parameters that were varied among simulations included site monthly mean climate, simulation length (yr) and, in certain cases, annual fire disturbance frequency and severity. Simulation length was set to the approximate age of each stand, which was assessed from tree cores (see above). Mean ages of stands by ecoregion varied from 100 yr near the coast to 300 yr in the western Cascades (Busing, 2004); the duration of most simulations was 200 to 600 yr. Species parameters were not varied among simulations.

Interpolated climate data were used for each site (Lugo et al., 1999). Nutrient supplies were set at adequate levels so as not to limit tree growth. Except for fire, large-scale natural disturbances were not employed. Small-scale disturbance in the form of canopy gap formation from endogenous tree mortality is a fundamental characteristic of the gap model. In this case, endogenous mortality refers to the death of trees simulated as a probabilistic function of maximum lifespan (background mortality) or in association with reduced annual tree growth (stress mortality). The primary abiotic environmental modifiers of tree growth were suboptimal temperatures, light availability, and drought stress.

The tests focused on eight forested ecoregions in western Oregon. Forested ecoregions (Omernik classification; Thorson et al., 2002) centered north of 43° N latitude and having 20 or more field data sites with old trees were used in the analysis (Fig. 1). In the Willamette Valley (ecoregion 3), where survey data on old stands were scarce, data from stands with trees approaching or exceeding 100 yr were collected to supplement the survey data set. For each ecoregion, measured basal area of dominant species, and of the entire stand served to quantify composition and stand dimensions. We used the selected sets of field plot sites (n=20, see above) within each ecoregion. Prior to the simulations, we determined that the selected field sites represented the larger pool of eligible field sites available in the ecoregions. We did so by examining means and distributions for measured variables describing stand composition and structure for each ecoregion. All analyses included only live trees >10 cm DBH.

Initial simulations were constrained by climate regime and age of the oldest tree (which dictated simulation length) at each randomly selected site. In additional simulations, regimes of fire disturbance were superimposed on the stands. Various regimes of fire, differing in disturbance frequency and

severity, characteristic of each ecoregion (USDA Fire Regime Database, J. Kertis, Siuslaw National Forest, personal communication) were simulated (Table 1). Fire return intervals ranged from 15 to 400 yr (frequency 0.067 to 0.0025 yr⁻¹). Fire regimes ranged from severe events with long return intervals (mean 400 yr) in ecoregions 1d and 4c to those dominated by low severity events with short return intervals (15 yr) in ecoregions 3, 9d and 9e.

Additional assumptions and procedures for the FORCLIM exercises followed Busing and Solomon (2004, 2005). Further details related to the fire disturbance simulations are presented herein (see Appendix).

Table 1. Mean fire return interval applied in the simulations by severity and ecoregion.

Ecoregion	Return interval of light fires (yr)	Return interval of severe fires (yr)
Coastal Volcanics (1d)	none	400
Mid-coastal Sedimentary (1g)	400	400
Willamette Valley (3)	25	300
West Cascade Lowlands (4a)	200	200
West Cascade Montane (4b)	300	300
Cascade Crest (4c)	none	400
East Cascade Ponderosa pine (9d)	15	400
East Cascade Pumice Plateau (9e)	15	400

Results and Discussion

Forest basal area

The simulations lacking fire disturbance gave total basal area values that greatly exceeded actual values. Simulated mean basal area was higher than the actual mean basal area in each of the eight ecoregions (Fig. 4a & b). When appropriate fire regimes were simulated for each ecoregion, the agreement between simulated and actual mean basal area improved markedly (Fig. 4c). The low values of total basal area in the Willamette Valley (ecoregion 3) were captured only by simulations with fire. Similarly, only the simulations including fire approximated the low total stand basal area of forest dominated by *Pinus ponderosa* in the Pumice Plateau (ecoregion 9e).

Forest composition

The data on actual vegetation showed a shift in tree species composition from *Pseudotsuga* dominated forests in coastal and montane regions of western Oregon (ecoregions 1-4) to *Pinus* dominated forests in central Oregon (ecoregion 9) (Fig. 4). An increase in subalpine conifers was evident at high

elevations of the Cascade Crest (ecoregion 4c). Simulations without fire produced forests dominated by *Pseudotsuga* in all eight ecoregions studied. The simulation clearly overestimated biomass in drier forests subject to high frequencies of low-severity fires (ecoregions 3, 9d & 9e). Simulations with high fire frequencies reduced the levels of *Pseudotsuga* and increased the relative dominance of very fire-tolerant species such as *Pinus ponderosa* and *Quercus garryana* in these ecoregions.

With the addition of fire to the simulations the only major discrepancy between actual and simulated composition is abundance of *Pinus ponderosa* in the simulated forests of the Willamette Valley (ecoregion 3). Although *Pinus ponderosa* occurs in the valley it is not currently a dominant species. It may have been more important prior to settlement and timber harvesting by Europeans (Johannessen et al., 1971). The past century of fire suppression in the valley (Robbins, 1997) may have inhibited *Pinus ponderosa* regeneration after initial losses to logging.

Other discrepancies include the presence of several minor species in the simulations that are rare or absent in actual forests. For example, *Abies procera*, *Picea sitchensis*, *Pinus ponderosa*, and *Quercus garryana* all occurred as minor species in certain simulated forests where they were not expected. Their presence reflects the fact that the factors limiting establishment and growth of tree species in the simulations are simplifications of reality. The most critical assumption in this regard was that species' propagules were available in all simulated stands. In reality they were probably available only in the vicinity of their rare parent trees, and hence, not consistently available at many sites.

Assumptions of climatic tolerances also may have played a role in the simulated presence of rare species. While all simulated dominant species within the ecoregions we examined were anticipated given the climatic conditions simulated, some discrepancies were noted for minor species. For example, *Picea sitchensis* appeared as a very minor species in some simulated montane forests. In reality, it is rare except along the coast. In this case, the simplified climatic regimes of the simulations do not fully account for coastal fog and they cannot restrict fog-belt species accordingly.

In conclusion, fire effects simulated as elevated mortality of trees based on their species and size do alter forest structure and composition. Low frequency fires (return interval >200 yr) have minor effects on composition. When they are severe, they tend to reduce total basal area with little regard to species differences. In the current study, simulated basal areas averaged across ecoregions were reduced to levels approximating actual basal areas. High frequency fires (return interval <30 yr) have major effects on species composition and on total basal area. They can cause substantial reductions in total basal area and shifts in dominance toward highly fire tolerant species.

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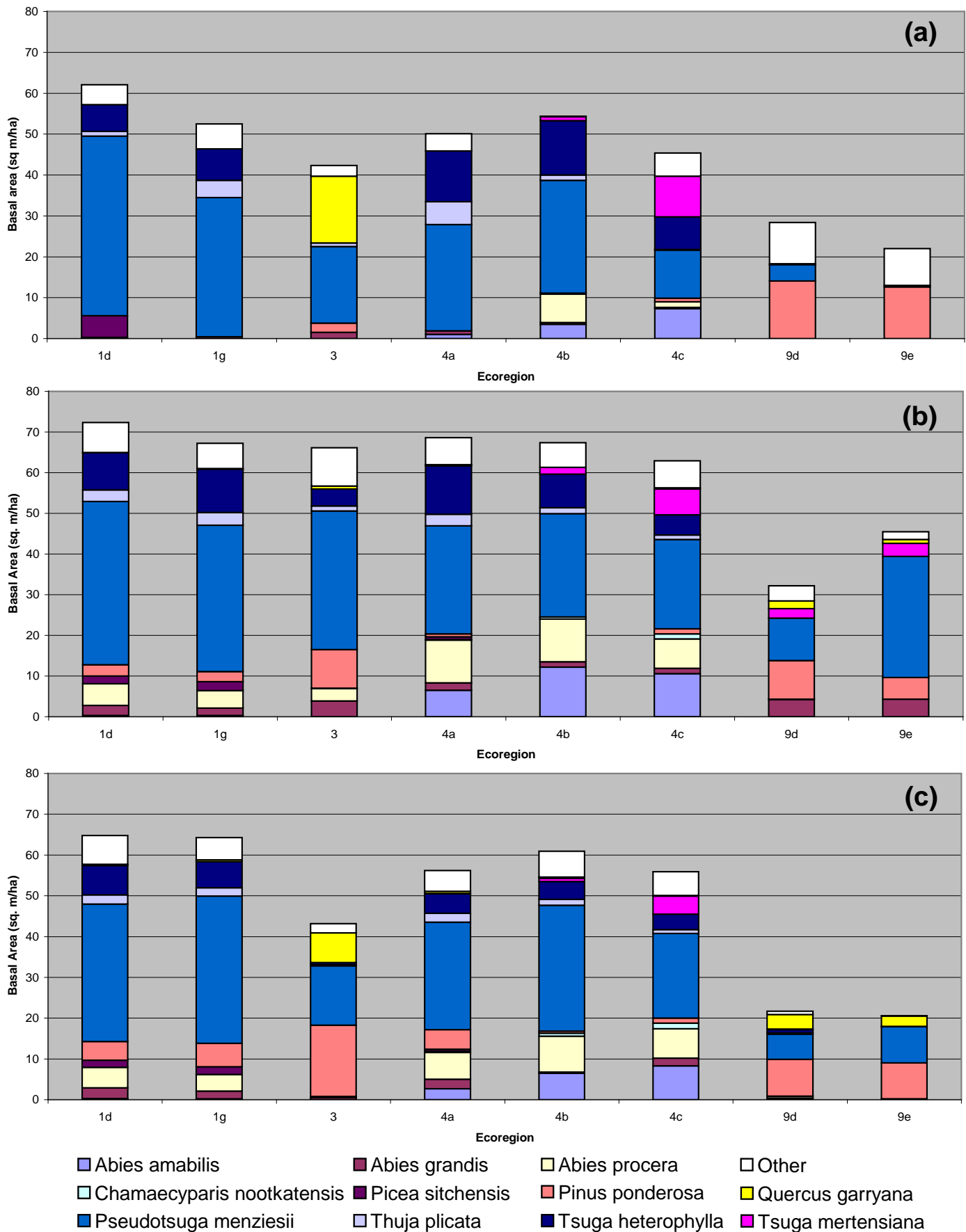


Figure 4. Graphs of mean species composition and basal area by ecoregion: a) actual data, b) simulated data, and c) simulated data with fire-induced mortality. Ecoregion codes are described in Figure 1.

Summary

The FORCLIM model of forest dynamics was applied to investigate the effects of various fire regimes on structure and composition of Pacific Northwest forests. Eight ecoregions in western Oregon with contrasting climate and fire regimes were studied. They ranged from wet coastal forests with infrequent fires to dry interior forests with frequent fires of low severity. The mesic coastal and montane forests were dominated by tall conifers, particularly *Pseudotsuga menziesii*. Dry forests of the interior were dominated by *Pinus ponderosa*. Fire disturbance simulated over centuries as elevated mortality of trees based on their species and size affected forest structure and composition. Fires of low frequency (return interval >200 yr) had minor effects on species composition, but severe fires reduced total basal area. The addition of severe fires often resulted in simulated average ecoregion basal areas closer to actual basal areas. High frequency fires (return interval <30 yr) had major effects on species composition and on total basal area. Even low-severity fires reduced total basal area and caused shifts in tree species composition when they were frequent.

Acknowledgements

Harald Bugmann and Rusty Dodson gave helpful advice on use of the FORCLIM model. Richard Hardt, Bob McNitt, and David Hibbs helped us find remnant forest vegetation in the Willamette Valley ecoregion. Don Phillips and Sarah Shafer provided constructive comments on earlier drafts of the manuscript. The field data used in comparisons were collected under ecological survey programs of the U.S. Department of Agriculture and the U.S. Department of the Interior.

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Appendix A

Implementation of fire disturbance simulations with FORCLIM 2.9

The 2004 version of FORCLIM with fire disturbance of mixed severity is described here. It is important to note that this simulator is based on the C program of model version 2.9. The C program was created by Rusty Dodson in 1996. It was converted from the MODULA2 program version 2.9, written by Harald Bugmann in the mid 1990s and documented by Bugmann and Solomon (2000). An earlier version of FORCLIM was documented by Bugmann (1994, 1996).

In the current study, fire disturbance was simulated as either a severe event or a low-severity event. Species were assigned to one of four fire tolerance classes. Fire tolerances were determined from Minore (1979), Dale and Hemstrom (1984), and the USDA Fire Effects Database.

The variable kFiT is the fire tolerance class, with class 1 being the least tolerant and most likely to be killed by fire. Values of all input variables, including fire tolerance, are provided below. Note that some values of drought tolerance (kDrT) and degree-day minimum (kDDMin) differ from those used by Bugmann circa 1996. These alterations were documented by Busing and Solomon (2004).

The simulator randomly determines in what year(s) a fire event occurs. The constants setting disturbance frequency are assigned in program module SET_CONSTANTS.C by the user. Two levels of severity are used in this exercise and each is simulated independently, with separate frequencies (or annual probabilities) and random number seeds for fire events. Low severity fire frequency is held in the constant kPGrFire; severe fire frequency is held in the constant kPFire. In a fire year an additional death probability is determined for each tree

based on the fire tolerance class of the species, fire severity, and tree diameter at breast height (dbhcm). In this investigation fire severity (fsev) is assumed to be 0.1 for low-severity fires and 1.0 for severe fires. Non-random interdependence of fire events among simulated plots or years is not considered here. More advanced fire regimes are possible through modification of the program code, however. The following equations are used to assign death probabilities (FireDP variable) to each individual in a fire year.

Fire tolerance equals 1:

$$\text{FireDP}=1.0$$

Fire tolerance equals 2:

$$\text{FireDP}=\exp(((1-fsev)*0.00202)+0.00053)*\text{dbhcm})$$

Fire tolerance equals 3:

$$\text{FireDP}=\exp(((1-fsev)*0.02745)+0.00255)*\text{dbhcm})$$

Fire tolerance equals 4:

$$\text{FireDP}=(\exp(-0.00053*\text{dbhcm}))-0.5+((1-fsev)*0.5)$$

The equations are for values of fsev >0 to 1. The resulting curves for two contrasting levels of fire severity are shown in Fig. 3. All tolerance classes other than class 1 exhibit a decrease in mortality probability with tree size. Equations are based on exponents from Dale and Hemstrom (1984) for severe fires. We have modified the equations of Dale and Hemstrom to accommodate 1) low-severity fires and 2) species with low susceptibility to fire mortality (class 4).

Table 2. The FORCLIM species parameter input file for the Pacific Northwest.

Parameters include: sType (phenology and growth form), kDm (max. diameter), kHm (max. height), kAm (max. age), kG (growth scaling constant), kDDMin (min. degree day sum), kWITN (min. temperature tolerated), kWITX (low temperature required), kDrT (drought tolerance), kFIT (flood tolerance), kNTol (nitrogen demand), kBrow (browse tolerance), kLy (light demand), kLa (shade tolerance), kLQ (litter quality), kImmYr (immigration threshold), and kFiT (fire tolerance). This table contains values modified or added after 2002 (Busing and Solomon, 2004).

Genus	Species	sType	kDm	kHm	kAm	kG	kDDMin	kWiTN	kWi-TX	kDrT	kFIT	kNTol	kBrow	kLy	kLa	kLQ	kImmYr	kFiT
Abies	amabilis	E5	200	75	600	340	390	-10	0	0.2	3	3	1	0.08	1	3	N	1
Abies	grandis	E5	225	76	300	357	600	-12	3	0.45	3	3	1	0.14	3	3	N	2
Abies	lasiocarpa	E5	80	40	300	359	300	N	-7	0.35	3	3	1	0.14	3	3	N	1
Abies	procera	E2	275	85	600	363	550	-7	3	0.25	3	3	1	0.22	7	3	N	2
Acer	macrophyllum	D1	120	28	300	280	705	-3	7	0.45	3	3	2	0.14	3	2	N	2
Alnus	rubra	D1	130	38	150	551	600	0	8	0.3	3	3	2	0.47	7	2	N	2
Arbutus	menziesii	E2	150	34	500	154	965	0	8	0.45	3	3	1	0.22	3	3	N	2
Chamaecyparis	nootkatensis	E5	370	53	3500	171	390	-14	-1	0.25	3	3	1	0.08	3	3	N	1
Picea	engelmannii	E5	244	55	600	211	400	N	-3	0.4	3	3	1	0.14	3	3	N	1
Picea	sitchensis	E5	500	90	800	374	1252	0	5	0.2	3	3	1	0.08	3	3	N	1
Pinus	contorta(c)	E3	50	10	500	447	1252	3	7	0.35	3	3	1	0.47	9	3	N	2
Pinus	contorta(l)	E3	213	46	600	226	524	-15	-9	0.35	3	3	1	0.47	9	3	N	2
Pinus	monticola	E3	200	75	600	359	589	-12	3	0.3	3	3	1	0.14	5	3	N	2
Pinus	ponderosa	E5	275	80	600	324	965	-12	8	0.55	3	3	1	0.47	7	3	N	4
Pseudotsuga	menziesii(g)	E3	250	54	700	403	633	-15	-5	0.4	3	3	1	0.22	7	3	N	3
Pseudotsuga	menziesii(m)	E3	425	117	1400	315	633	-10	5	0.4	3	3	1	0.22	7	3	N	3
Quercus	garryana	D2	250	37	500	161	677	-4	6	0.5	3	3	2	0.47	5	2	N	4
Thuja	plicata	E5	350	76	1500	282	748	-8	3	0.25	3	3	1	0.08	3	3	N	2
Tsuga	heterophylla	E5	275	80	700	351	719	-8	4	0.25	3	3	1	0.08	1	3	N	1
Tsuga	mertensiana	E3	150	46	800	203	300	-15	-3	0.35	3	3	1	0.08	1	3	N	1